# ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
# BIOLOGICAL OPINION

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

**Activity:** Endangered Species Act Section 7 Consultation on the Continued Implementation of Management Measures for the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass Fisheries [Consultation No. F/NER/2012/01956]

**Consulting Agency:** National Marine Fisheries Service, Northeast Regional Office, through its Protected Resources Division

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## 1.0 Introduction

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

The NMFS NERO SFD has reinitiated formal intra-service consultation on the continued operation of seven fisheries as authorized by NMFS under their respective Fishery Management Plans (FMPs) issued under the authority of the Magnuson-Stevens Fisheries Conservation Act (MSA) and implementing regulations. Fisheries considered here are the: (1) Northeast multispecies (multispecies), (2) monkfish, (3) spiny dogfish, (4) Atlantic bluefish (bluefish), (5) Northeast skate complex (skate), (6) Atlantic mackerel/squid/butterfish (MSB), and (7) summer flounder/scup/black sea bass (FSB) fisheries (collectively referred to as "the seven fisheries" hereinafter). As described fully in section 2.2 below, reinitiation of these consultations is necessary as these fisheries may affect five distinct population segments (DPS) of Atlantic sturgeon that were listed as threatened or endangered on February 6, 2012 (77 FR 5880-5912; 77 FR 5914-5982). This document represents our biological opinion (Opinion) on the continued operation of these fisheries and the effects of their continued operation on ESA-listed species and designated critical habitat under our jurisdiction in accordance with section 7 of the ESA. See Section 3.0, Description of the Proposed Action, for a discussion on the rationale for the inclusion of these seven fisheries in a single consultation.

Formal intra-service section 7 consultation on the continued operation of the seven fisheries was reinitiated on February 9, 2012 [Consultation No. F/NER/2012/01956]. This Opinion is based on the information developed by NMFS NERO and other sources of information, as cited in the Literature Cited section of this document.

## 2.0 Consultation History

### 2.1 Consultations Review

In addition to the formal consultations outlined below for each of the seven fisheries, the effects of a variety of Amendments, Framework Adjustments (Frameworks), and other management measures were evaluated to determine if reinitiation had been triggered. All actions that did not trigger reinitiation of ESA consultation are not specifically discussed in the consultation histories of the seven fisheries below.

### 2.1.1 Multispecies

The consultation history for the multispecies fishery was reviewed in the previous formal consultation completed October 29, 2010. Briefly, the first formal consultation on the multispecies fishery was completed on June 12, 1986, and concluded that that operation of the fishery would not result in jeopardy to any ESA-listed species under NMFS jurisdiction. Consultation was reinitiated in response to the proposed implementation of Amendment 5 to the NE Multispecies FMP. That consultation was completed on November 30, 1993, and concluded that the continued operation of the multispecies fishery, including implementation of the Amendment 5 measures, would not jeopardize the continued existence of any listed species under NMFS jurisdiction.

In response to further changes to the management of the multispecies fishery under the NE Multispecies FMP, formal consultation was reinitiated and subsequently completed on February 16, 1996, and again on December 13, 1996. The December 1996 consultation concluded that the continued operation of the multispecies fishery would jeopardize the continued existence of right whales. An interim Reasonable and Prudent Alternative (RPA) to avoid the likelihood of jeopardy to right whales was provided with the Opinion. Consultation was reinitiated in 1997 to assess the effects of the NE Multispecies FMP's Framework Adjustment 23 that would implement a gillnet prohibition in the federal portion of Cape Cod Bay Right Whale Critical Habitat and in the Great South Channel, as specified in the RPA of the December 1996 Opinion. NMFS concluded that the proposed action was not likely to jeopardize the continued existence of the right whale, or other listed species, or result in adverse modification to right whale critical habitat. Later in 1997, consultation was reinitiated concurrent with the initial formal consultation on the Atlantic Large Whale Take Reduction Plan (ALWTRP). That consultation concluded that the continued operation of the multispecies fishery would not jeopardize any ESA-listed species under NMFS jurisdiction given that implementation of the ALWTRP, in conjunction with simultaneous right whale recovery actions taken by NMFS and other agencies, was expected to reduce the threat of entanglement for right whales in gillnet gear in the multispecies fishery.

In 1999, a right whale mortality was attributed to entanglement in gillnet gear. NMFS was unable to determine the origin of the gillnet gear (fishery in which the gear was being fished). In addition, other entanglements of right whales in gillnet gear were reported after completion of the 1997 Opinion. 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 multispecies fishery. Nevertheless, NMFS concluded that the entanglements did provide new information that revealed the action (the continued operation of the multispecies fishery) may affect listed species in a manner or to an extent not previously considered. Therefore, consultation was reinitiated. That consultation was completed on June 14, 2001 and concluded that the continued operation of the multispecies fishery, including measures previously implemented as part of the ALWTRP, was likely to jeopardize the continued existence of right whales. The RPA included with that opinion required the creation of a Seasonal Area Management (SAM) program and a Dynamic Area Management (DAM) program, both implemented as part of the revised ALWTRP.

On October 5, 2007, NMFS published a final rule in the *Federal Register* (72 FR 57104; October 5, 2007) that made many changes to the ALWTRP, including a change in the use of fixed gillnet gear in the multispecies fishery. As part of the final rule, the DAM program was eliminated as of April 7, 2008 and the SAM program was eliminated as of October 6, 2008.[1] The changes to the ALWTRP, therefore, modified the RPA in a manner that caused an effect to listed species not considered in the June 14, 2001 Opinion for the fishery. In accordance with 50 CFR 402.16, NMFS reinitiated formal consultation on the multispecies fishery on April 2, 2008 to reconsider the effects of the continued operation of the multispecies fishery on ESA-listed cetaceans and sea turtles. Additionally, in 2006, the Northeast Fisheries Science Center (NEFSC) released reference document 06-19 (Murray 2006) that reported on the annual estimated taking of loggerhead sea turtles in bottom-otter trawl gear fished in Mid-Atlantic waters during the period of 1996-2004. As a follow-up, and in response to a request from NERO, the bycatch rate identified in Murray (2006) was used to estimate the take of loggerhead sea turtles in all fisheries (by FMP group) using bottom otter trawl gear fished in Mid-Atlantic waters during the period of 2000-2004 (Murray 2008). Based on the approach as described in Murray (2008), the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear for the period of 2000-2004 was estimated to be 43 for trawl gear used in the Northeast multispecies fishery. NMFS also received an estimate of loggerhead sea turtle bycatch in sink gillnet gear from the NEFSC in November 2009 (Murray 2009a). In that report, the average annual bycatch of loggerhead sea turtles in sink gillnet gear potentially used in the multispecies fishery, coded in the report as "other species," was estimated to be three for the period of 2002-2006 (Murray 2009a). Because these bycatch estimates revealed effects of the multispecies fishery on sea turtles that were not previously considered in the June 14, 2001 Opinion, formal consultation was reinitiated. That consultation was completed on October 29, 2010, and concluded that the continued operation of the multispecies fishery was not likely to jeopardize the continued existence of any ESA-listed species.

### 2.1.2 Monkfish

The consultation history for the monkfish fishery was reviewed in previous formal consultations completed April 14, 2003 [Consultation number F/NER/2002/00196] and October, 29, 2010 [Consultation number F/NER/2008/01754]. In brief, formal consultation on the fishery was first initiated in 1998 and concluded that the operation of the fishery, including modification of the gillnet portion of the fishery as required under the ALWTRP, would not result in jeopardy to any ESA-listed species under NMFS jurisdiction. The Opinion also concluded that the gillnet sector might adversely affect sea turtles, and an Incidental Take Statement (ITS) with Reasonable and Prudent Measures (RPMs) to minimize take was provided. Consultation was reinitiated in 2000 after new information indicated a change in the status of right whales, and observer data indicated that the ITS for sea turtles in the monkfish fishery was exceeded during Year 1 (November 8, 1999-April 30, 2000) of the Opinion. The consultation [Consultation number F/NER/2001/00546] was concluded on June 14, 2001, and resulted in a jeopardy finding for

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

North Atlantic right whales. The Opinion contained one RPA with multiple management components that were designed to avoid the likelihood of the federal monkfish fishery jeopardizing the continued existence of the endangered right whale. Incidental take of sea turtles was also anticipated but was not expected to jeopardize any affected sea turtle species. An ITS was provided, along with RPMs, to minimize the taking of sea turtles in the monkfish fishery.

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

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

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

Due to changes in the ALWTRP, which eliminated the DAM program as of April 7, 2008, and the SAM program as of October 6, 2008,[2] and new information about the monkfish fishery's effects on sea turtle takes, formal consultation was reinitiated on April 2, 2008 to reconsider the effects of the continued operation of the monkfish fishery on ESA-listed cetaceans and sea turtles. That consultation was completed on October 29, 2010, and concluded that the continued operation of the monkfish fishery was not likely to jeopardize the existence of any ESA-listed species.

### 2.1.3 Spiny Dogfish

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

The consultation history for the spiny dogfish fishery was reviewed in a previous formal consultation completed October 29, 2010. Briefly, the Spiny Dogfish FMP was developed jointly by the Mid-Atlantic Fishery Management Council (MAFMC) and the New England Fishery Management Council (NEFMC) to eliminate overfishing and rebuild the stock of spiny dogfish. Prior to 1999, landings of spiny dogfish were managed under the Multispecies FMP. The effects of fisheries targeting spiny dogfish on listed species were therefore considered within the broad scope of fisheries prosecuted under the Multispecies FMP.

The first formal consultation on the spiny dogfish fishery was completed on August 13, 1999, and concluded that operation of the fishery would not result in jeopardy to any ESA-listed species under NMFS jurisdiction. For endangered whales, this conclusion was based on the assumption that the incorporation of measures identified in the ALWTRP into the Spiny Dogfish FMP would be effective at reducing incidental mortality and serious injury of the whales. This conclusion was also based on NMFS' December 13, 1996 Opinion that identified implementation of the ALWTRP as an effective RPA to avoid the likelihood of jeopardy for fisheries managed under the Multispecies FMP.

In 1999, a right whale mortality was attributed to entanglement in gillnet gear. NMFS was unable to determine the origin of the gillnet gear (fishery in which the gear was being fished). In addition, other entanglements of right whales in gillnet gear were reported in the same time period. 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 the spiny dogfish fishery. Nevertheless, NMFS concluded that the entanglements did provide new information that the action (the continued operation of the spiny dogfish fishery) may affect listed species in a manner or to an extent not previously considered. Therefore, consultation was reinitiated on May 4, 2000. That consultation was completed on June 14, 2001, and concluded that the continued operation of the spiny dogfish fishery, including measures previously implemented as part of the ALWTRP, was likely to jeopardize the continued existence of right whales. The RPA included with that Opinion required the creation of the SAM and DAM implemented as part of the revised ALWTRP.

On October 5, 2007, NMFS published a final rule in the *Federal Register* (72 FR 57104; October 5, 2007) that made many changes to the ALWTRP, including a change in the use of fixed gillnet gear in the spiny dogfish fishery. As part of the final rule, the DAM program was eliminated as of April 7, 2008 and the SAM program was eliminated as of October 6, 2008.[3] The changes to the ALWTRP, therefore, modified the RPA in a manner that caused an effect to listed species not considered in the June 14, 2001 Opinion for the fishery. NMFS reinitiated formal consultation on the spiny dogfish fishery on April 2, 2008 to reconsider the effects of the continued operation of the spiny dogfish fishery on ESA-listed cetaceans and sea turtles. That consultation was completed on October 29, 2010, and concluded that the continued operation of the spiny dogfish fishery was not likely to jeopardize the existence of any ESA-listed species.

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

### 2.1.4 Bluefish

The consultation history for the bluefish fishery was reviewed by NMFS in a previous formal consultation completed on October 29, 2010. Briefly, the Bluefish FMP was developed in the 1980s, and was the first FMP to be jointly developed by an interstate commission and a Regional Fishery Management Council. Currently, bluefish is jointly managed by the MAFMC and ASMFC. Amendment 1 to the FMP was considered in a 1999 Opinion, in which NMFS concluded that the continued operation of the bluefish fishery 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 1999). However, sea turtles and shortnose sturgeon were expected to experience harassment, injury, or mortality due to interactions with the gear associated with this fishery. An ITS was provided with the 1999 Opinion along with non-discretionary RPMs to minimize the impacts of incidental take.

In 2010, new information on large whale interactions and sea turtle bycatch in net gear consistent with that used in the bluefish fishery triggered reinitiation. The 2010 Opinion issued by NMFS concluded that the continued operation of the bluefish fishery would not jeopardize the existence of right, humpback, fin, and sei whales, or loggerhead, leatherback, Kemp's ridley, and green sea turtles, nor was it likely to destroy or adversely modify right whale critical habitat (NMFS 2010a). However, ESA-listed large whales and sea turtles 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. An ITS for sea turtles was issued along with the 2010 Opinion. The ITS exempted the annual incidental take of up to three loggerheads over a five-year average in trawl gear, of which up to two per year may be lethal, and up to 79 loggerheads annually over a five-year average in gillnet gear, of which up to 32 per year may be lethal. For the other three sea turtle species, lethal or non-lethal takes of up to four leatherback, four Kemp's ridley, and five green sea turtles in trawl and gillnet gear combined are exempted annually. RPMs and accompanying terms and conditions to minimize the impacts of incidental take were also provided in the ITS (NMFS 2010a).

### 2.1.5 Skates

The implementation of the Skate FMP was first reviewed by NMFS in a formal consultation initiated on March 12, 2003 and completed on July 24, 2003. The 2003 Opinion issued by NMFS concluded that the initial implementation of the Skate FMP would not jeopardize the continued existence of right, humpback, fin, sei, blue, and sperm whales, or loggerhead, leatherback, Kemp's ridley, and green sea turtles, and was not likely to adversely modify right whale critical habitat (NMFS 2003b). An ITS was provided with the 2003 Opinion along with non-discretionary RPMs to minimize the impacts of incidental take. As described in the ITS, up to one loggerhead, leatherback, Kemp's ridley, or green sea turtle (one turtle only of any of these four species) was anticipated to be injured or killed annually as a result of the implementation of the Skate FMP.

NMFS next considered the effects of the continued operation of the skate fishery under the Skate FMP on ESA-listed marine mammals, sea turtles, and fish during formal section 7 consultation initiated on April 2, 2008. An Opinion resulting from this consultation was completed on October 29, 2010. It concluded that the continued implementation of the Skate FMP, including Amendment 3 (which was enacted in July 2010), may adversely affect, but would not jeopardize the continued existence of right, humpback, fin, and sei whales, or loggerhead, leatherback, Kemp's ridley, and green sea turtles, nor would it destroy or adversely modify designated right whale critical habitat. An ITS for sea turtles was issued along with the Opinion exempting the annual incidental take of up to 24 loggerheads over a five-year average in trawl gear, of which up to 11 per year may be lethal, and up to 15 loggerheads annually over a five-year average in gillnet gear, of which up to six per year may be lethal. For the other three sea turtle species, lethal or non-lethal takes of up to four leatherback, four Kemp's ridley, and five green sea turtles in trawl and gillnet gear combined are exempted annually. Non-discretionary RPMs to minimize the impacts of incidental take were also provided in the ITS (NMFS 2010e).

NMFS has also informally reviewed a number of frameworks, amendments, exempted fishing permits, and emergency actions associated with the Skate FMP over the past several years. These reviews have concluded that either the proposed actions may affect, but were not likely to adversely affect, ESA-listed species or designated critical habitat under NMFS jurisdiction or that the proposed actions did not trigger reinitiation of formal section 7 consultation.

#### 2.1.6 Atlantic Mackerel/Squid/Butterfish

The first formal consultation on the MSB fishery was conducted in the context of the consultation on all fisheries for the Marine Mammal Exemption Program (MMEP). An Opinion with an ITS for marine mammals in all commercial fisheries was issued on July 5, 1990. Subsequently, NMFS completed informal consultations for Amendment 4 (August 6, 1991), Amendment 5 (February 16, 1995), and Amendment 6 (August 15, 1995) to the FMP. Due to the low level of incidental take of endangered or threatened species in the fishery, formal consultation was not initiated for this fishery independently of the MMEP consultation and no separate ITS was issued.

The second formal consultation was triggered when NMFS became aware of possible sea turtle interactions by vessels targeting mackerel and/or squid while considering Amendment 8 actions. A formal consultation on the MSB fishery was conducted during the normal regulatory review process to implement Amendment 8 on the FMP, and the Opinion was completed April 28, 1999, with an ITS. The MSB fishery continued under this ITS until 2010, when consultation on the FMP was reinitiated due to new sea turtle bycatch information.

The 2010 Opinion issued by NMFS concluded that the continued operation of the MSB fishery would not jeopardize the continued existence of right, humpback, fin, and sei whales, or loggerhead, leatherback, Kemp's ridley, and green sea turtles, nor was it likely to destroy or adversely modify right whale critical habitat (NMFS 2010b). However, ESA-listed sea turtles were expected to experience harassment, injury, or mortality due to interactions with the gear associated with this fishery. Interactions between these species and MSB fishing gear can include captures or entanglements in net gear and, on rarer occasions, hooking (internally or externally) or entanglements in hook and line gear. An ITS for sea turtles was issued along with the Opinion, which also included RPMs and accompanying terms and conditions to minimize the impacts of incidental take (NMFS 2010b).

In addition to these formal consultations, informal section 7 consultations were conducted and completed for Amendment 9 and Amendment 10 (2009). The most recent informal consultation occurred in 2011 for Amendment 11, which established a cap on capacity in the mackerel fishery via a limited access program and established an allocation for the recreational mackerel fishery to facilitate implementation of Annual Catch Limits (ACL) and Accountability Measures (AM). The 2009 and 2011 informal consultations concluded that the proposed amendments either had no effect on or might affect, but were not likely to adversely affect, ESA-listed species under NMFS jurisdiction or designated critical habitat.

#### 2.1.7 Summer Flounder/Scup/Black Sea Bass

The first formal consultation on a Summer Flounder FMP concluded in 1988 that operation of this fishery would not jeopardize any ESA-listed species under NMFS jurisdiction. Consultation was reinitiated in 1990 following documented sea turtle takes in the summer flounder fishery, and a new Opinion concluded in August 1991 that operation of the summer flounder trawl fishery was likely to result in jeopardy for Kemp's ridley sea turtles. The Opinion included Reasonable and Prudent Alternatives of restricted tow times for bottom trawlers and the establishment of a monitoring program, and indicated that additional conservation measures, such as the use of TEDs or closure of the fishery, would be imposed if necessary. In December 1991, NMFS implemented an emergency rule requiring the use of Turtle Excluder Devices (TEDs) in the summer flounder trawl fishery operating off North Carolina and southern Virginia waters (56 FR 234, December 5, 1991). Formal consultation for the proposed inclusion of the scup and black sea bass fisheries in the FMP concluded on February 24, 1996, that operation of these fisheries, as well as the continued operation of the summer flounder fishery, was not likely to jeopardize the existence of listed species and would not result in the destruction or adverse modification of designated critical habitat.

In 2001, increased landing limits were proposed for each fishery for the 2002 fishing year. Given that increases in landing limits can result in increases in effort, NMFS reinitiated consultation on the FMP in 2001 to consider the effects of the proposed action on ESA-listed species and designated critical habitat within the management area. The December 16, 2001 Opinion for the FSB fishery concluded that continued operation of the fishery was not likely to jeopardize the continued existence of listed species and would not result in the destruction or adverse modification of designated critical habitat. An ITS was provided in the Opinion that described the anticipated annual take (lethal or non-lethal) in trawl, gillnet, and trap/pot gear used in the fishery.

The next formal consultation was completed on October 29, 2010. The 2010 Opinion issued by NMFS concluded that the continued operation of the FSB fishery would not jeopardize the continued existence of right, humpback, fin, and sei whales, or loggerhead, leatherback, Kemp's ridley, and green sea turtles, nor was it likely to destroy or adversely modify right whale critical habitat (NMFS 2010g). However, ESA-listed sea turtles were expected to experience harassment, injury, or mortality due to interactions with the gear associated with this fishery. Interactions between these species and FSB fishing gear can include captures or entanglements in net and pot/trap gear and, on rarer occasions, hooking (internally or externally) or entanglements in hook and line gear. An ITS for sea turtles was issued along with the Opinion, along with RPMs and accompanying terms and conditions to minimize the impacts of incidental take (NMFS 2010g).

### 2.2 Cause for Reinitiating

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

On February 6, 2012, NMFS issued two final rules (77 FR 5880-5912; 77 FR 5914-5982) listing five DPSs of Atlantic sturgeon as threatened or endangered. Four DPSs (New York Bight, Chesapeake Bay, Carolina, and South Atlantic) are listed as endangered and one DPS (Gulf of Maine) is listed as threatened. The effective date of the listing was April 6, 2012. We have reinitiated formal section 7 consultation on the seven fisheries because we determined that the newly listed Atlantic sturgeon DPSs may be affected by this action (Memo to the Record, D. Morris, February 9, 2012).

## 3.0 Description of the Proposed Action

The proposed action is the continued operation of the following seven fisheries: Northeast multispecies, monkfish, spiny dogfish, Atlantic bluefish, northeast skate complex, mackerel/squid/butterfish, and summer flounder/scup/black sea bass.

The traditional approach to conducting section 7 consultations on agency actions related to commercial fisheries has been to address each FMP as a separate federal action. In previous biological opinions, we have estimated take and analyzed impacts to ESA-listed species for each FMP individually. Often, the takes and impacts are then broken down by gear type for more specific analysis. While this approach is useful for loggerhead sea turtles, for which NEFSC has been able to provide us with bycatch estimates by FMP and gear type, we have concluded that it is not feasible to apply the same protocol to Atlantic sturgeon.

In spring 2011, we requested that the NEFSC conduct a bycatch analysis for Atlantic sturgeon by FMP, similar to what NEFSC has provided in the past for loggerhead sea turtles. NEFSC responded that the primary causes of Atlantic sturgeon takes are deployments of particular gear types in specific areas and time periods, and that the partitioning of discard encounters to specific FMPs is not a particularly informative exercise because of the high likelihood of inappropriately attributing associations/responsibilities. Nevertheless, on February 16, 2012, we sent NEFSC a memorandum proposing a method to reallocate the Atlantic sturgeon bycatch estimate contained in the August 2011 NEFSC report to match Atlantic sturgeon bycatch with fishing effort and the appropriate FMP. The proposed re-allocation did not change the overall bycatch estimate, but distributed it among FMPs. On March 2, 2012, NEFSC replied that they were unable to endorse our methodology,[4] and that they had continued reservations about the utility and credibility of attributing sturgeon takes to FMPs.

Due to the likely inaccuracies of attributing Atlantic sturgeon takes to any particular FMP, we decided to examine the relevant FMPs as a "batch," in one consultation, examining Atlantic sturgeon interaction with seven fisheries by gear types. The seven fisheries included in this Opinion use the two types of gear, sink gillnets and bottom otter trawls, which are known to interact with Atlantic sturgeon, as well as trap/pot gear and longlines, which are not known to interact with Atlantic sturgeon. Examining these seven FMPs comprehensively allows a more useful analysis, and ensures that bycatch reduction measures are not placed erroneously on a particular FMP (*i.e.*, a fishery could undergo an additional regulatory or management burden, but not yield corresponding benefit in reduction of Atlantic sturgeon take if apportionment is incorrect). This Opinion considers the effects of the above-listed FMPs on Atlantic sturgeon, as well as on NMFS ESA-listed sea turtles, whales, and Atlantic salmon. For loggerhead sea turtles, we evaluate the impacts by FMP and gear type, and the loggerhead ITS will reflect the allocation of take by FMP. For other species of sea turtles and for whales, prior biological opinions have analyzed the effects on the species by gear type rather than FMP, with the same analysis repeated for each individual FMP. In this Opinion, we follow that precedent and analyze the effects on non-loggerhead sea turtles and whales, as well as on Atlantic sturgeon and Atlantic salmon, by gear types.

For the purposes of this analysis, the following FMPs are not included in this batch due to either no recorded interactions or very low numbers of interactions expected between Atlantic sturgeon and the gear deployed to catch target species under the FMPs: tilefish, deep sea red crab, surf clam/ocean quahog, and herring. Atlantic sea scallops and American lobster were each considered in biological opinions dated July 12, 2012 and August 3, 2012, respectively. Due to the unique nature of the fishing gears used and the very low number of interactions, NMFS determined that reinitiation was not triggered for tilefish, herring, red crab and surf clam/ocean quahog FMPs as the newly listed Atlantic sturgeon are not likely to be affected by these actions.

Recently, stock assessments and essential fish habitat analyses for the seven fisheries have been conducted at five-year intervals. Due to frequent changes in the seven fisheries, habitat, and status of the resources, using stock and Essential Fish Habitat (EFH) assessments to inform management decisions beyond five years is not realistic. Our time frames for producing new bycatch estimates for loggerheads and Atlantic sturgeon in trawl, gillnet, and dredge fisheries

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[4] Our proposed methodology was similar to the method of Warden (2010), where takes were attributed to FMPs in accordance with landings composition. The NEFSC stated that their analysis did not find evidence that Atlantic sturgeon take on a trip was proportional to the total catch of a FMP. The NEFSC went on to say that the application of the Warden method led to inappropriate conclusions about the FMP associations due to the rarity of Atlantic sturgeon combined with the heterogeneity of fishing activities within each gear/area/year strata.

occur on staggered five-year cycles, with additional periods of time to assess whether there have been significant changes in bycatch rates from one time period to the next. Large whale stock assessment reports also analyze data in five year intervals. Therefore, taking into account the different timelines for all these assessments and the likelihood of new data, we expect that we will have to evaluate whether there is a need to reinitiate consultation on the seven fisheries at some point in the next ten years, and that beyond ten years the effects of the seven fisheries in combination with environmental changes on ESA-listed species may be quite different than they are currently.

Given the time frames related to the data on which management of the seven fisheries are based, we do not believe that it is possible to reliably analyze effects of the action far into the future. Anticipating that the seven fisheries will operate the same way for more than ten years is not only speculative, but the history and pace of change in the fisheries described in sections 2.0 and 3.0 suggests that it is not reasonable to expect the seven fisheries to continue to operate as they do currently beyond ten years from now. Longer-term effects of the seven fisheries on ESA-listed species, whatever they may be, are more difficult to pinpoint and extrapolate beyond ten years. Since the distribution of effort in the seven fisheries and the status of the resource can change over just a few years,we have determined to limit the scope of the action assessed in this Opinion to ten years. However, our analysis of effects does consider impacts of these actions within this ten year time frame that may extend beyond the 10 year time frame. 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.

## 3.1 Description of the Gear Used in the Seven Fisheries

Sink gillnets and bottom otter trawls are the two predominant gear types used in the seven fisheries. However, trap/pot gear is often used in the black sea bass and scup fisheries and accounted for a significant portion (approximately 46%) of the landings in the black sea bass fishery since 1998 (NEFSC 2012). Hook gear (*i.e.*, handlines and bottom longline) is also used in the seven fisheries. The use of other gear types (*e.g.*, pound nets, mid-water and paired trawls, haul and purse seines, troll and rod and reel) occurs at much lower levels and is not discussed further in this effects analysis because usage within these fisheries is so low that we don't believe they will have any effects on the listed species.

**Sink gillnets** are panels of net, 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.

**Bottom trawls** are typically cone-shaped nets towed on the bottom. Large, rectangular doors attached to the two cables keep the net open while deployed. At the bottom of the 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 centimeters in diameter) rubber discs or steel bobbins (rockhoppers) that ride over structures such as boulders and coral heads that might otherwise snag the net. Some bottom trawls are constructed with tickler chains that disturb the seabed to flush shrimp or fish species into the water column to be caught by the net. The constricted posterior netting of a bottom trawl which retains the catch is called the cod-end.

**Trap/pot gear** consists of the trap, buoy/surface line, groundline, buoys, and/or highflyers. The traps are baited and rest on the bottom until the trap is retrieved. Buoy line(s) connect to the trap and rise vertically to the surface. Traps may be set singly with each trap having its own surface line and buoy, or may be fished in trawls consisting of two or more traps per trawl. Multiple traps are linked together by sinking groundline, with at least one, but most often two surface lines and buoys. The surface lines are typically at an end of a series of traps to mark the location of the gear. Fish pots and hand lines are generally fished in inshore waters and target black sea bass (with the exception of some lobster and sea bass targets in NY) (NEFSC 2012). Trap gear configuration in state waters is more similar from state to state than different. However, depending on the coastal topography, some states may have a higher percentage of single traps/pots in the water versus trawls when compared to others. Offshore gear includes additional line at or near the surface that connects a radar reflector highflyer to one of the buoys to aid in relocation and "visibility" of the gear. Excess buoy line is restricted from floating at the surface and all buoys, flotation devices, and/or weights must be attached to the buoy with a weak link. Per the ALWTRP regulations, all trap/pot gear is required to be hauled out of the water at least once every 30 days and fishermen are encouraged, but not required, to maintain knot-free buoy lines.

**Bottom longlines** are a series of groundlines connected to a flag(s) and marker buoy(s) by a buoy line. Each ground line has many gangions attached, which are generally nylon braids to which a baited hooks are secured. An anchor holds the groundline in place. The groundline is allowed to "soak" on the bottom for a number of hours until the fisherman considers it appropriate to pull in the groundline and remove the hooked fish.

## 3.2 Description of the Current NE Multispecies Fishery

The proposed action includes the continued operation of the multispecies fishery managed under the Northeast (NE) Multispecies FMP including measures implemented by Amendment 16 as well as Framework Adjustments 44-50. Also included as part of the proposed action are recently implemented changes as outlined by Amendment 19, the sectors operation plans final rules for 2012 and 2013, and a cod emergency action. The proposed final rule for the small mesh fishery will also be included as part of the proposed action.

The Multispecies FMP has a long management history, which is briefly summarized here. In 1977, the NEFMC issued an interim Northeast Multispecies FMP, which implemented a quota-based system for three species: cod, haddock, and yellowtail flounder. The FMP did not limit entry into the fishery, which resulted in increasing participation, and a "race to catch" the allowable quota. The quota system was eliminated in 1982, and replaced with other management measures, including minimum fish size, cod-end mesh regulations, and closed areas to protect spawning haddock (NEFMC 2009a).

During the late 1980s and early 1990s, four amendments were added to the plan to further manage the stocks of large-mesh species, but these amendments did not prevent overfishing. In 1986, a new NE Multispecies FMP was implemented, which set species mortality targets based on calculated maximum spawning potential. It also expanded the number of species included in the management unit. Management measures included minimum sizes, spawning closed areas, and reduced areas and time periods for small-mesh fishing in the GOM.

In 1994, Amendment 5 established a moratorium on new vessel permits during the rebuilding period (creating the current limited access permit system based on history in the fishery), implemented a DAS effort reduction program, added mesh size restrictions, included interim gillnet regulations to reduce harbor porpoise bycatch, established a mandatory vessel trip reporting system for landings, prohibited pair-trawling, changed some minimum fish sizes, and expanded the size of Closed Area II. Shortly after, Amendment 6 implemented additional haddock conservation measures.

In 1996, Amendment 7 accelerated the DAS effort reduction program, eliminated exemptions from the effort control program, provided incentives to use larger mesh than the minimum size, broadened the area closures to protect juvenile and spawning fish, increased the haddock possession limit, established rebuilding programs, changed existing permit categories, and created a program for reviewing the management measures annually and making changes to the regulations through the framework adjustment process. Amendments 8-12 and several Framework Adjustments were then added to achieve the Amendment 7 fishing mortality targets and to fulfill the requirement for annual adjustments to management measures (NEFMC 2009a).

The NEFMC began work on Amendment 13 in February 1999 to address the need to develop rebuilding programs and to address problems identified with the DAS effort control program. In the meantime, the NEFMC implemented Framework Adjustment 33 to meet the Amendment 7 requirement for an annual adjustment to the FMP on May 1, 2000.

In 2004, Amendment 13 established three DAS categories (A, B, and C), established the Closed Area II Yellowtail Flounder SAP [5], allowed sectors of the groundfish fishing industry to develop sector allocation plans, undertook an adaptive approach for rebuilding groundfish stocks, and

[5] There are three SAPs currently in place: The Closed Area I Hook Gear Haddock SAP is open to NE multispecies DAS vessels fishing with hook gear in a portion of Closed Area I; the Eastern U.S./Canada Haddock SAP is open to NE multispecies DAS vessels using a haddock "separator" trawl in portions of the Eastern U.S./Canada Area and Closed Area II; and the Closed Area II Yellowtail Flounder/Haddock SAP is open to multispecies DAS vessels fishing for yellowtail flounder or haddock in the southern portion of Closed Area II. Only Closed Area II Yellowtail Flounder SAP was implemented under Amendment 13.

implemented several provisions of the U.S./Canada Resource Sharing Understanding (NEFMC 2009a).[6]

After the adoption of Amendment 13, four Framework Adjustment actions (Frameworks 40A, 40B, 41, and 42) followed. Frameworks 40A, 40B, and 41 implemented several measures and programs to provide opportunities for vessels to target healthy groundfish stocks to mitigate some of the social and economic impacts of management measures implemented through Amendment 13. Some of the changes included creating a Georges Bank (GB) yellowtail flounder rebuilding strategy, changes in trap limits, changing DAS counting, establishing the GB Cod Fixed Gear Sector, extending the DAS leasing program, modifying the DAS transfer program, requiring installation of a Vessel Monitoring System (VMS) for all limited access DAS groundfish vessels, and changing gear standards.

**Amendment 16 Final Rule (2010)**
In May 2010, the Amendment 16 final rule implemented a broad range of measures designed to achieve mortality targets for species managed by the NE Multispecies FMP, make substantial changes to sector management, and implement Magnuson-Stevens Fishery Conservation and Management Act (MSA) requirements regarding the establishment of ACLs and AMs. Amendment 16 also was implemented in order to provide opportunities to target healthy stocks, mitigate (to the extent possible) the economic impacts of the measures, and improve administration of the fishery. New status determination criteria developed by the NEFSC during its 2008 assessment were adopted, as were control rules for setting Acceptable Biological Catch (ABC) and methods for calculating and distributing ACLs among fisheries that catch groundfish stocks. Revisions to mortality targets to achieve rebuilding based on the recent stock assessments were implemented. Formal rebuilding programs were implemented for witch flounder, winter flounder (Georges Bank), pollock, northern windowpane flounder, and Atlantic wolffish.

**Sector Operation Plans Final Rules**
Amendment 16 authorized 17 new sectors throughout the New England region. Sectors are self-selecting and largely self-regulating. The FMP rules regarding sector measures were extensively revised, including measures supporting sector implementation, methods for drafting and submitting formation proposals, operations plans, sector monitoring plans, enforcement provisions, and the interaction of sectors with special management programs. Under the Amendment 16 measures, sectors conduct fishing activity according to their own operations plans that must be annually approved by NMFS. Sectors are allocated a certain amount of the ABC for each groundfish stock based upon the sum of the proportional landings histories for each of the permits that joined the sector in each fishing year (FY). This allocation is known as a sector's annual catch entitlement (ACE) for each stock. Once a sector catches its ACE for any stock, the sector must stop fishing in the stock area associated with that species for the remainder of the FY, or until it acquires additional ACE for that stock from another sector. In order to

[6] The U.S./Canada Resource Sharing Understanding (Understanding) was reached between the United States and Canada regarding the management of GB cod, GB haddock, and GGB yellowtail flounder resources found within the waters of both countries in an area known as the U.S./Canada Management Area. Amendment 13 implements certain measures consistent with the Understanding, including a requirement to use a VMS, an area declaration requirement, and specific gear requirements (flatfish net or haddock separator trawl).

assure that sector catch limits are not exceeded, a new system monitoring catch, including at-sea and dockside catch monitoring, was implemented.

Vessels that are not participating in a sector for a particular fishing year (known as common pool vessels) continue to be subject to existing effort controls, including DAS allocations, trip limits, area closures, size limits, and gear restrictions. Common pool vessels are charged DAS in 24-hour increments.

**Annual Approval of Sector Operations Plans**
On an annual basis, each sector submits an operations plan to NMFS that specifies participants in the sector, outlines expected operations, and requests exemptions from existing regulations. Sectors receive exemptions from many of the common pool effort control measures in exchange for fishing under a quota system where they are limited to a specific amount for each stock (i.e., the ACE described above). In FYs 2010, 2011, and 2012 sectors received exemptions from various measures, including trip limits, certain rolling closures, seasonal DAS restrictions, and gear restrictions. Using FY 2012 as an example, sectors requested additional exemptions from gear restrictions, special access programs measures, and minimum fish size limits. Specifically, sectors requested to be allowed to use a Ruhle trawl without rockhoppers when using a flat sweep, to haul another vessel's hook gear, to access the Closed Area II Yellowtail Flounder/Haddock Special Access Program (SAP) and the Eastern U.S./Canada Haddock SAP earlier in the FY, to fish inside and outside of the Closed Area I Hook Gear Haddock SAP on the same trip, and to land headed haddock, among other administrative provisions. The final rule for FY 2013 sector operations plans (except for certain exemptions and measures in proposed operations plans) was approved and implemented on May 1, 2013.

**Framework Adjustment 44 (Final Rule–2010)**
Effective in May 2010, Framework Adjustment 44 implemented multi-year catch specifications for the fishery, and modified effort control measures to achieve mortality targets as follows. The measures with changes to management of the fishery include:

- *ABCs and ACL Specifications:* ABCs and ACLs were adopted implemented for each managed stock for FYs 2010 through 2012, based upon the methods implemented by Amendment 16 that take into account biological and management uncertainty, and based upon the best available science.

- *Commercial Fishery Effort Control Modification:* Effort control measures for common pool vessels were modified because of uncertainty over future sector membership and the possibility that fishing behavior may change in ways not predicted by the analytic tools used to develop Amendment 16. To address this latter concern, the NMFS Regional Administrator was provided with the authority to modify common pool effort control measures, including possession limits and DAS counting rates, at any time during the year to increase the likelihood that ACLs will be met and not exceeded.

**Framework Adjustment 45 (Final Rule–2011)**
Framework Adjustment 45, implemented in May 2011, revised the biological reference points and stock status for pollock, updated ACLs for several stocks for FYs 2011–2012, adjusted the

rebuilding program for Georges Bank (GB) yellowtail flounder, increased scallop vessel access to the Great South Channel Exemption Area, approved five new sectors, modified the existing dockside and at-sea monitoring requirements, revised several sector administrative provisions, established a Gulf of Maine (GOM) Cod Spawning Protection Area, refined measures affecting the operations of NE multispecies vessels fishing with handgear, and approved the FY 2011 U.S./Canada Management Area total allowable catches (TACs).

**Framework Adjustment 46 (Final Rule–2011)**
Framework Adjustment 46, which became effective September 14, 2011, was developed to address haddock catch in the Atlantic herring fishery. The rule increases the haddock incidental catch cap allocated to the Atlantic midwater trawl herring fishery to 1% of the GB haddock ABC and to 1% of the GOM haddock ABC. In addition, this action modified the AMs applicable to the Atlantic herring fishery such that, upon reaching the haddock incidental catch cap, the midwater trawl herring fleet could not catch or land herring in excess of the incidental catch limit (2,000 lb/907.2 kg) in or from the appropriate haddock stock area. This action is intended to allow the herring fishery to fully use available herring quota, while providing incentives for the midwater trawl fishery to minimize haddock bycatch.

**Framework Adjustment 47 (Final Rule–2012)**
Framework Adjustment 47 to the NE Multispecies FMP became effective in May 2012. This action: 1) revised the status determination criteria for three winter flounder stocks and Gulf of Maine cod; 2) revised the GB yellowtail flounder rebuilding strategy; 3) changed the administration of the scallop fishery's yellowtail flounder ACLs; 4) adopted acceptable biological catches and ACLs for FY 2012–2014 for 10 stocks; 5) removed the cap that limits scallop vessel catch of yellowtail flounder in the GB access areas; 6) eliminated the restricted gear areas for common pool vessels adopted in Amendment 16; 7) adopted a zero-possession proactive AM for Southern New England/Mid-Atlantic winter flounder and Atlantic wolfish; 8) adopted area-based AMs for both windowpane flounder stocks and ocean pout; and 9) prohibited possession of Atlantic halibut if the ACL is exceeded.

**Framework Adjustment 48 and 50 (Final Rules–2013)**
Framework Adjustment 48 and 50 to the NE Multispecies FMP became effective in May 2013. These measures include: catch limits for FYs 2013-2015 for many of the groundfish stocks, including FY 2013 TACs for U.S./Canada stocks of Eastern GB cod and haddock; new and revised catch limits and AMs for certain fisheries; a revised Southern New England/Mid-Atlantic (SNE/MA) winter flounder rebuilding program and allowance of SNE/MA winter flounder landings; and reductions in the minimum fish size for some species, among other measures.

**GOM Cod Emergency Action (Final Rule–2012)**
NMFS prepared a supplemental EA for Framework 47 to revise recreational GOM cod fishery measures for FY 2012. This action revised measures to reduce mortality resulting from the recreational fishery. The action reduced the minimum fish size for cod caught by recreational and charter party vessels in the GOM Regulated Mesh Area from 24 inches to 19 inches, and reduced the associated possession limits for both private recreational and charter/party vessels to nine fish per angler per day. The action made no revision to the existing seasonal GOM cod possession prohibition.

**Small Mesh Management**

The management of the small-mesh NE multispecies fishery began in 1991, when Amendment 4 incorporated silver and red hake into the FMP, and established an experimental fishery on Cultivator Shoal. Framework Adjustment 6 (1994) increased the minimum mesh size from 2.5 inches to 3 inches. Small Mesh Areas I and II, off the coast of New Hampshire, were established in Framework Adjustment 9 (1995). The NEFMC established essential fish habitat (EFH) designations and added offshore hake to the plan in Amendment 12 (2000). Also in Amendment 12, the Council proposed limited entry into the small mesh fishery. However, that measure was not approved by the Secretary of Commerce and has not been implemented to date, although efforts are underway to reconsider limiting entry into this fishery. The Raised Footrope Trawl Area off of Cape Cod was established in Framework Adjustment 35 (2000). Framework Adjustments 35 and 37 modified and streamlined some of the varying management measures to increase consistency across the exemption areas, Framework Adjustment 38 established the Grate Raised Footrope Exemption Area in the inshore GOM area.

**Small-Mesh Secretarial Amendment (Final Rule–May 2012)**

NMFS prepared a Secretarial Amendment to the NE Multispecies FMP to implement ACLs and AMs for the small-mesh multispecies fishery (silver hake, red hake, and offshore hake, only) prior to the start of the 2012 fishing year. The Secretarial Amendment only established ACLs and AMs, and is not expected to modify any of the management measures, including the exemption programs and trip limits. NEFMC’s amendment (Amendment 19) to implement ACLs and AMs replace the measures in the Secretarial Amendment. Amendment 19 also modified other aspects of the small-mesh multispecies fishery, including trip limits for both red hake and silver hake.

There are only a few stocks in the NE Multispecies FMP that have a notable recreational component. For those stocks, the FMP addresses the recreational component on a stock-by-stock basis, as necessary. The principal recreational species landed have been cod, haddock, and winter flounder, with some pollock and insubstantial amounts of other stocks. With the implementation of Amendment 16 to the NE Multispecies FMP and the setting of discrete catch levels for all stocks in the large mesh fishery, the ABC has been distributed among the various components of fisheries that operate in the Northeast in order to account for various sources of catch. This proportion of catch allocated to the recreational fishery and to state and federal waters depends upon the particular stock. For GOM cod and haddock, there is a discrete recreational allocation, whereas for other stocks, no such allocation is made due to the relatively minor amount of recreational catch. For some stocks, an allocation for state waters is made to account for anticipated recreational catch. The overall split of the ACL of GOM cod between commercial and recreational fisheries was determined by the NEFMC based on historical catch (34% recreational; 66% commercial). The amount of recreational harvest of cod from state waters (without regard to stock) averaged 19% from 2001 to 2008, but was highly variable and ranged from 9% to 35%. For GOM haddock, the overall split of the annual catch limit between commercial and recreational fisheries was set at 73% and 27%, respectively. For GOM winter flounder, the recreational fishery occurs almost entirely in state waters, and the FMP set aside 25% of the ABC for state waters to account for this fishery. For SNE/MA winter flounder, the FMP set aside 8% of the ABC to account for recreational catch in state waters. This was increased to almost 30% through Framework 47 to account for increased catches in state waters.

About half of the pollock recreational catch has been from state waters since 2001. Although currently there is no allocation of pollock for the federal recreational fishery, the FMP may incorporate such an allocation in the future to facilitate accountability.

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. In order to better understand the impacts of recreational fishing on sea turtles, in 2012 NMFS initiated a survey-based pilot study planned to continue through 2013. This pilot study will assess the extent of interactions between recreational anglers and sea turtles, and includes shore-based, private vessel, and charter/headboat fishing effort. The pilot study for this work has been conducted in the southeast Atlantic states. Therefore, NMFS is unable to estimate an amount or extent of take occurring in the recreational component of the multispicies fishery at this time and will instead focus the majority of the effects analysis on the commercial component of the fishery.

Sixteen species of groundfish are managed under the NE Multispecies FMP. Thirteen species (20 stocks) are managed as part of the large-mesh complex, based on fish size and type of gear used to harvest the fish (Atlantic cod, haddock, pollock, yellowtail flounder, witch flounder, winter flounder, windowpane flounder, American plaice, Atlantic halibut, redfish, ocean pout, white hake, and Atlantic wolfish). Three species (silver hake/whiting, red hake, and offshore hake) are included in the FMP as the small-mesh complex, but are managed under a separate small-mesh multispecies program through a series of exemptions to the NE Multispecies FMP. Although large-mesh and small-mesh species are managed under the same FMP, they are effectively managed as two different fisheries. For example, Amendment 16 to the FMP implemented ACLs and AMs for the large-mesh species and ocean pout, and the Small-Mesh Secretarial Amendment to the FMP implemented such measures for the remaining small-mesh species. These small-mesh groundfish species exhibit unique body types, behaviors, and habitat preferences, but all are demersal (live near the bottom and feed on benthic organisms). Groundfish are found throughout New England waters, from the GOM to southern New England.

There are a variety of fishing gears used in the multispecies fishery. Authorized fishing gear includes gillnet, trawl, longline, hook and line, trap/pot, dredge, seine, and spear (FR 50 600.725(v)). Trap/pot, dredge, seine, and spear gear will not be discussed in this Opinion due to the negligible amount of NE multispecies landed by these gear types. Otter trawls are the primary gear type used for all species in both the large-mesh and small-mesh complexes, and flatfish and silver hake are caught almost exclusively with otter trawls. Recreational fishing for groundfish is focused primarily on Atlantic cod, pollock, haddock, red hake, and winter flounder. Recreational fishing is conducted by shore-based anglers and anglers with private boats, as well as by anglers aboard party/charter vessels (NEFMC 2009a).

Between FY 2001 and FY 2011, bottom trawls and sink gillnets accounted for a large majority of total landings of large-mesh groundfish species in each year, as shown in Table 1 (Vessel Trip Report Database). Bottom trawls accounted for the majority of large-mesh species landings. Total bottom trawl landings of large-mesh species declined from a high of 82 million pounds in FY 2001 to a low of nearly 36 million pounds in FY 2006. Since FY 2006, bottom trawl landings have increased to between 42 and 49 million pounds. Between 2000 and 2009, bottom trawls accounted for the overwhelming majority of small-mesh multispecies landings. Total landings, including total bottom trawl landings, of small-mesh multispecies have declined throughout that time period (Table 2).

Sink gillnets landed the second highest percentage of groundfish. The amount of groundfish landings by gillnets has been relatively consistent between FY 2001 and FY 2011. However, the percentage of total and groundfish landings by gillnets has increased during the FY 2001-FY 2011 period to a high of 24.8% in FY 2008. This increase in percentage within the fishery is a result of the decrease in total landings and trawl gear landings as opposed to an increase in gillnet fishing effort.

**Table 1** Large-mesh groundfish landings (in pounds) by vessels targeting groundfish and other fisheries vessels not targeting groundfish, by gear used, FY 2001-FY 2011(Vessel Trip Report Database, NMFS).

| Gear Type | FY 2001 | FY 2002 | FY 2003 | FY 2004 | FY 2005 | FY 2006 | FY 2007 | FY 2008 | FY 2009 | FY 2010 | FY 2011 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Bottom Trawl | 82,073,862 | 68,026,014 | 64,971,047 | 60,067,134 | 49,539,300 | 35,907,933 | 42,298,968 | 46,507,678 | 46,028,534 | 44,915,226 | 48,681,134 |
| Sink Gillnet | 12,608,484 | 9,258,642 | 11,393,380 | 10,117,279 | 9,545,181 | 10,044,465 | 13,135,199 | 15,871,290 | 14,513,326 | 7,822,410 | 8,709,422 |
| Bottom Longline | 2,625,847 | 1,227,172 | 1,134,194 | 2,229,028 | 2,722,202 | 1,435,194 | 1,083,690 | 1,193,491 | 1,337,258 | 1,086,560 | 1,028,484 |
| Handline | 1,971,009 | 776,646 | 389,794 | 415,512 | 192,714 | 242,391 | 197,351 | 297,391 | 395,894 | 111,409 | 205,290 |
| *Scallop Dredge | 91,371 | 9,252 | 13,355 | 41,337 | 10,935 | 19,018 | 24,553 | 9,071 | 10,745 | 7,447 | 27,772 |
| *Lobster Trap | 35,767 | 17,671 | 11,941 | 9,208 | 7,080 | 2,913 | 4,232 | 7,919 | 1,904 | 3,322 | 1,602 |
| *Midwater Trawl | CONF | | | | CONF | 1,575 | 50,385 | 8,600 | 34,989 | 87,143 | 36,808 |
| *Shrimp Trawl | 3,971 | 1,572 | 2,204 | 80 | CONF | 2,575 | CONF | 1,411 | 748 | 288 | 12,468 |
| *Other | 95,926 | 126,799 | 122,136 | 85,140 | 36,839 | 16,408 | 21,526 | 5,238 | 60,153 | 14,715 | 8,270 |
| Grand Total | 99,506,237 | 79,443,768 | 78,038,051 | 72,964,718 | 62,054,251 | 47,672,472 | 56,815,904 | 63,902,089 | 62,383,551 | 54,048,520 | 58,711,250 |

*Not targeting groundfish

"Conf" indicates landings comprised of fewer than three vessels that must be kept confidential pursuant to 50 CFR 600.425(a).

**Table 2** Small-Mesh Multispecies Landings in pounds, by gear used, FY 2000-2009 (Vessel Trip Report Database, NMFS)

| Gear Type | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 |
|---|---|---|---|---|---|---|---|---|---|---|
| Longline | 2,821 | 2,749 | 786 | 948 | 2,029 | 133,324 | 66,215 | 471 | 42,538 | 343 |
| Bottom Trawl | 30,436,422 | 31,868,082 | 19,463,105 | 20,739,681 | 19,431,217 | 16,299,264 | 12,471,451 | 14,546,567 | 13,695,305 | 13,688,528 |
| Shrimp Trawl | 48,502 | 2,205 | 9 | | 397 | 305 | 2,205 | 2,205 | 6,614 | 17,637 |
| Sink Gillnet | 119,050 | 83,776 | 57,552 | 64,910 | 170,394 | 149,207 | 187,784 | 190,111 | 412,312 | 811,461 |
| Other | 213,848 | 233,690 | 60,074 | 108,027 | 679,652 | 974,207 | 589,087 | 494,557 | 1,080,242 | 4,085,529 |
| Total | 30,820,643 | 32,190,502 | 19,581,526 | 20,913,566 | 20,283,690 | 17,556,307 | 13,316,742 | 15,233,912 | 15,237,012 | 18,603,499 |

During the period of FY 2001-2006 commercial landings of large-mesh multispecies declined from 99,506,237 pounds to 58,711,250 pounds (NEFMC 2011). Commercial landings of small-mesh multispecies likewise declined, from 32,149,000 pounds to 18,603,499 pounds during the same time period (NEFMC 2011). Such declines are believed to have been, at least in part, due to changes in management of the multispecies fishery. Information on the history of the fishery with respect to management measures was provided in the January 2011 Environmental Assessment for Framework Adjustment 45 (NEFMC 2011).

For management purposes, the fishing year for the multispecies fishery is defined as May 1 through April 30. The multispecies fishery is managed by the NEFMC using a variety of management tools, including DAS, special management programs, area closures, gear requirements, trip limits, and sectors. The vast majority of the fishery's active vessels in 2010 and 2011 fished under sector management rather than the common pool. The GOM and GB small-mesh fishery is managed using seasonal mesh size exemption programs, and operates year-round as a gear-exempted fishery in Southern New England and the Mid-Atlantic Exemption Areas. NMFS NERO administers the management program for the multispecies fishery under the authority specified in the MSA. While NMFS may independently enact management measures, most management measures for the multispecies fishery are developed through a participatory regulatory process conducted by the NEFMC. NEFMC actions are reviewed by the Secretary of Commerce, and implemented by NMFS if found consistent with all legal requirements.

### 3.3 Description of the Current Monkfish Fishery

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

Although there is no strong evidence of separate biological stocks, monkfish are divided into two distinct management areas, analogous to two distinct stock areas, to accommodate differences in fishery practices. The Northern Fishery Management Area (NFMA) includes waters from Maine to Cape Cod, MA, and the Southern Fishery Management Area (SFMA) includes waters from Cape Cod to North Carolina. There is no known recreational fishery for monkfish, but they are sometimes taken by anglers fishing for other bottom-dwelling fish. The monkfish fishery is jointly managed by the NEFMC and the Mid-Atlantic Fishery Management Council (MAFMC), with the NEFMC having the administrative lead. During the early 1990s, commercial fishermen and dealers in the monkfish fishery raised several issues regarding monkfish to both the New England and Mid-Atlantic Councils ("the Councils"), including concerns about the increasing amount of small fish being landed commercially, the increasing frequency of gear conflicts between monkfish vessels and those in other fisheries, and the expanding directed monkfish trawl fishery. In response, the Councils developed the joint FMP that was implemented in 1999. For the first eight years under the FMP, the fishery was in a rebuilding plan since the stocks were considered overfished (below the biomass target). The FMP was designed to stop overfishing and rebuild the stocks through a number of measures, including: limiting the number of vessels with access to the fishery and allocating DAS to those vessels; trip limits for vessels fishing for monkfish; minimum fish size limits; gear restrictions; mandatory time out of the fishery during the spawning season; and a framework adjustment process to develop or revise management measures based on changing conditions in the fishery.

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

The two gears predominantly used in the directed monkfish fishery are bottom trawls and bottom gillnets. Trawl gear accounts for most of the reported landings in the NFMA (73% during 2000-2011), while gillnets account for the majority of the landings in the SFMA (72% during 2000-2011). During 2000-2011, 46% of all reported U.S. monkfish landings were taken in otter trawls, 5% in sea scallop dredges, 48% in gillnets, and 0.21% in other gear (NMFS Analysis and Program Support Division data as of October 19, 2012). Dredges, spears, and hook gear are minor components of effort and landings in this fishery. Monkfish trawl fishing in the NFMA is often conducted in conjunction with Northeast multispecies fishing, while gillnets are used in the SFMA directed monkfish fishery. Because the vast majority of directed effort and landings for monkfish occurs with bottom trawls and gillnets, this Opinion will primarily focus on potential effects from these gear types.

Dealer-reported total landings (live weight) increased from an average of 2,500 metric tons in the 1970s to 8,700 metric tons in the 1980s, 23,000 metric tons in the 1990s (50th Northeast Regional Stock Assessment Review Committee 2010). From 2000 to 2005, dealer reported total monkfish landings averaged 22,000 metric tons, dropping to 10,000 metric tons during 2006-2011 (NMFS/NERO/APSD). Reported total landings in 2011 were 3,699 metric tons in the NFMA and 5,801 metric tons in the SFMA, a slight increase in landings from both management areas compared to landings during fishing years 2009 and 2010, but less than half of the landings reported in fishing year 2003. Overall, total landings have declined since 2003 due to management regulations, including TACs of 5,000 metric tons in the NFMA and 5,100 metric tons in the SFMA during 2007-2010. Monkfish TACs have since been raised to 8,925 metric tons and 5,854 metric tons in the NFMA and SFMA, respectively, for fishing years 2011-2013, suggesting that landings have the capacity to continue to increase in the upcoming years. Landings in the early part of the time series are thought to be under-reported. The accuracy of landings data has likely improved with mandatory reporting beginning in 1994.

Amendment 1 to the Monkfish FMP, enacted April 1999, implemented the EFH provisions of the Magnuson-Stevens Act. Amendment 2, which was implemented in May 2005, included restrictions on otter trawls in certain areas, made the minimum fish size consistent in all areas, closed two offshore canyons to monkfish fishing, created a monkfish research DAS set-aside program, and created new permit categories for fishing in designated areas, among other measures. In 2007, the Councils proposed Framework 4 to set catch targets (TTACs) at 5,000 metric tons and 5,100 metric tons for the NFMA and SFMA, respectively. In 2007, the Northeast Data Poor Stocks Working Group (DPWG) completed a monkfish stock assessment, recommending revisions to the biomass reference points. The Councils requested the DPWG to evaluate the impact of applying those TTACs for the 2007-2009 fishing years. The DPWG concluded that, under those catch targets, fishing mortality rates would remain below the threshold and biomass would continue an upward trend that would take it above the biomass target. Upon receiving the DPWG report, NMFS approved Framework 4, including an automatic extension of the TTACs beyond FY 2009 if the Councils did not adopt new targets. The Councils adopted the new reference points as Framework Adjustment 5 to the Monkfish FMP, which were then implemented in May 2008.

In 2007, the MSA was revised to include, among other things, the requirement that all FMPs establish ACLs and AMs. For stocks not subject to overfishing, such as monkfish, the MSA set a deadline of 2011 for the implementation of ACLs and AMs. In 2009, NMFS published revised National Standard 1 Guidelines, which the Councils have used to develop ACLs and AMs for all FMPs.

In May 2011, Amendment 5 implemented the MSA-mandated ACLs and AMs and specified DAS and corresponding trip limits for the monkfish fishery. Amendment 5 also modified the Research Set Aside Program, implemented a provision to minimize bycatch resulting from trip limit overages, and enabled vessels to land monkfish heads separate from the bodies. However, in 2010, after the Council submitted Amendment 5, the 50$^{th}$ Stock Assessment Review Committee (SARC 50) completed a new monkfish stock assessment, declaring that neither stock of monkfish are considered overfished, and that overfishing is not occurring on either stock. Due to the newly available science, the DAS and trip limit specifications for the NFMA were disapproved in Amendment 5. To address the disapproved measures, Framework Adjustment 7 set the specifications for the NFMA and adopted new biomass reference points for both areas based upon the newly available science from SARC 50.

In late 2010, the Councils began the development of Amendment 6 to the FMP that is considering implementing a form of catch shares in the monkfish fishery. The Councils held a series of public meetings on catch shares soliciting public comment through March 7, 2011. Amendment 6 is still being developed by the Councils and is not expected to be implemented until at least FY 2014.

### 3.4 Description of the Current Spiny Dogfish Fishery

Spiny dogfish range from Labrador to Florida, although they are most abundant from Nova Scotia to Cape Hatteras, NC. They migrate seasonally, moving north in spring and summer, and south in fall and winter. Canadian research surveys indicate that spiny dogfish are distributed throughout the Canadian Maritimes during the summer months. The stock is concentrated in U.S. waters during the fall through spring.

Spiny dogfish are known to consume a wide variety of prey, including ctenophores, squid, hake, sand lance, mackerel, herring, flatfish, and sculpins, as well as jellyfish, crabs, octopods, and sea cucumbers. In spite of their large numbers and opportunistic feeding, spiny dogfish, like many elasmobranches, suffer from several reproductive constraints. Females may take 7-12 years to reach maturity, growing more than one-third larger than their mature male counterparts before becoming sexually mature. Fertilization and egg development are internal, and gestation takes roughly two years, resulting in litters that usually average six to seven dogfish "pups." As a result of these factors (long time to maturity, long gestation periods, and low fecundity), spiny dogfish are vulnerable to overfishing, particularly if fishing activities focus on the largest individuals, which are almost all mature females.

As a result of increased fishing pressure, spiny dogfish were classified as overfished in 1998. The Councils jointly developed an FMP for spiny dogfish. This plan was partially approved in 1999 and implemented in 2000. Management measures included an overall commercial quota allocated into two semiannual periods; restrictive trip limits (600 lbs); a prohibition on finning; an annual quota adjustment process; and permit and reporting requirements. The most significant effect of the measures was the elimination of the directed dogfish fishery in federal waters. Framework Adjustment 1 to the FMP, implemented in January 2006, provided for a multi-year, rather than annual, specification-setting process. The 2006 assessment of the dogfish stock found the stock no longer overfished, but not rebuilt (43rd Northeast Regional Stock Assessment Workshop (NEFSC 2006).

Most spiny dogfish landings are the result of commercial fishing activities, as reported recreational landings comprise less than 2% of the total catch. Because of the restrictive commercial trip limits designed to restrict the directed dogfish fishery, dogfish landings predominantly occurred as bycatch from other commercial fisheries, although the increased quotas and possession limits (3,000 lbs per trip) in the May 1-April 30 fishing years (FYs) 2009, 2010, and 2011 have resulted in a small-scale directed fishery. Sink gillnets, bottom longlines, and bottom otter trawls are the primary commercial fishing gears that catch spiny dogfish and these three gear types accounted for 97% of all dogfish landed between 2006 and 2011 (NMFS/NERO/APSD). Spiny dogfish landings came mostly from sink gillnets (69.7%) and otter trawl (19.0%), but some landings consistently come from longline (8.45) and handline (2.7%). From FYs 2000 through 2008, the federal FMP allowed for a 4 million pound quota two-season fishery with 57.9% of the quota being allocated to Period 1 (May 1 through October 31), and 42.1% to Period 2 (November 1 through April 30). The trip limit for both periods was 600 pounds/trip. Commercial landings ranged from 5.1 million pounds in fishing year 2001 to as low as 1.5 million pounds in 2004, and increased to 22.5 million pounds in 2011 as the stock rebuilt. The majority of commercial landings are made in Massachusetts ports.

In state waters, zero to three nautical miles (nm) from shore, spiny dogfish are managed under the Atlantic States Marine Fisheries Commission (ASMFC) Interstate FMP for Spiny Dogfish (implemented in 2003). Spiny dogfish management in state waters under the Interstate FMP deviated from the federal FMP in 2003, 2006-2008, 2010, and 2011. In 2006 through 2008, due to an increase abundance of spiny dogfish, states increased the coastwide quota while the federal quota remained the same. However, in 2010 and 2011, the state quotas were slightly lower than the federal quota due to quota overage deductions from the previous year. While the quota for both interstate and federal FMPs has varied in past years, both FMPs are intended to cover the entire spiny dogfish population along the

Atlantic coast of the United States (i.e., in both state and federal waters from 0-200 nm).

In the fall of 2009, the Northeast Fisheries Science Center (NEFSC) updated the spiny dogfish stock status using the model from the 43rd SARC, 2008 catch data, and results from the 2009 trawl survey. Based on the scientific findings, NMFS declared that the spiny dogfish stock was not overfished and overfishing was not occurring. For FY2009, state and federal quotas were set consistently at 12 million pounds with 3,000 pound trip limits.

Framework Adjustment 2 (Framework 2) to the FMP, enacted July 24, 2009, provided for automatic incorporation of biological reference points into the FMP as they become recommended through peer-reviewed assessments. The spiny dogfish stock was formally declared rebuilt in June 2010, after new scientific information providing an official biomass target became available. As a result, the FY2010 quota slightly increased from that of FY 2009 and was set at 15 million pounds with 3,000 pound trip limits. Through the procedure outlined in Framework 2, the 2010 spiny dogfish specifications updated the Spiny Dogfish FMP to incorporate the new biomass reference point values. For FY 2011, state and federal quotas were set consistently at 20 million pounds, with a 3,000 pound trip limit. The relatively low trip limits are believed to discourage a large-scale directed fishery for spiny dogfish.

The 2012 spiny dogfish fishery specifications were implemented on June 21, 2012 (77 FR 30224). The specifications were designed to establish an annual catch limit, commercial quota, and trip limits for the spiny dogfish fishery. The FY 2012 commercial quota (35.694 million lb) implemented in state and Federal waters represents a 78% increases from the FY 2011 level (20 million lb). However, proportionate trip limit increases were not implemented. These measures were enacted to help avoid fishery closures, prolong the fishing season, and reduce regulatory discards of spiny dogfish during the 2012 fishing year.

Specifications for the Spiny Dogfish FMP were implemented at the beginning of the 2013 fishing year to cover annual specifications for 2013-2015. The specifications were produced to establish annual catch limits, commercial quotas and possession limits for the spiny dogfish fishery.

Effective June 1, 2013, an exempted fishery for vessels fishing with a NE Federal spiny dogfish permit in two separate areas, were established as follows: When using gillnet and longline gear from June through December, and handgear from June through August, in an area east of Cape Cod, MA; and when using longline gear and handgear from June through August in an area west of Cape Cod, MA. The areas of this exempted fishery will be referred to as the Eastern and Western Cape Cod Spiny Dogfish Exemption Areas. Vessels participating in this exempted spiny dogfish fishery that hold a Federalspiny dogfish permit may land up to 4,000 lb of spiny dogfish per trip outside the confines of the NE multispecies regulations.

Vessels will be limited by the spiny dogfish annual quota, which is divided into two seasons.

### 3.5 Description of the Current Atlantic Bluefish Fishery

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 the following documents: 2013 and 2014 Bluefish Specifications, Environmental Assessment, and Initial Regulatory Flexibility Analysis (MAFMC 2013); Status of Fishery Resources off the Northeastern US – Bluefish (Shepherd 2006); the revised 41st Northeast Regional Stock Assessment Workshop (41st SAW) Assessment Report (NEFSC 2006); Bluefish 2012 Stock Assessment Update (Wood 2013); 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 EFH 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 2006; Shepherd and Packer 2006). Bluefish are described as warm-water migrants and usually do not occur in Mid-Atlantic Bight[7] 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[8] and southern New England, and move south in fall-winter to waters in the South Atlantic Bight[9] 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, FL, 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 MAFMC and the ASMFC and implemented in 1990. The management measures presently include an overall annual landings quota in which

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

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

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

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 2006; Shepherd 2006). This has routinely occurred over the past several years and is proposed to occur again in 2013 and 2014, as the commercial fishery will be allocated an increased percentage of the total quota (around 37%-38% of the total allowable landings; 9.076 million lbs in 2013, 8.674 million lbs in 2014) 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 68% of the total bluefish landings in 2011 (MAFMC 2013). 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 per day (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 state-specific restrictions on bluefish possession limits and recreational size limits.

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 2013). 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 2013). Allocations of the bluefish quota are not equally divided among 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 (MAFMC 2013). Florida receives approximately 10% of the annual quota, but has fully harvested its quota share in recent years (MAFMC 2013). Gillnets account for the vast majority of bluefish landed in the commercial fishery. In 2011, gillnets accounted for 93.4% of the directed catch of bluefish, while hook gear accounted for 4.5% and other gear categories caught the remaining 2.1% (MAFMC 2013). Aside from gillnets, gear types authorized for use in the commercial bluefish fishery include trawl, longline, handline, bandit, rod and reel, pot, trap, seine, and dredge gear (50 CFR 600.725(v)).

### 3.6 Description of the Current NE Skate Complex Fishery

The current management measures for the skate fishery, the history of the fishery, and the general distribution and habitat preferences of skates are described in the Final Fishery Management Plan for the Northeast Skate Complex (NEFMC 2003), Final Environmental Impact Statement with an Initial Regulatory Flexibility

Analysis for Final Amendment 3 to the Fishery Management Plan for the Northeast Skate Complex (NEFMC 2009b), Status of Fishery Resources off the Northeastern US – Skates (Sosebee 2006), and revised 44th Northeast Regional Stock Assessment Workshop (44th SAW) Assessment Report (NEFSC 2007a). Additional information on the distribution and habitat characteristics of skates can be found in the EFH source documents for the seven skate species in the Northeast Region (Packer *et al.* 2003a, 2003b, 2003c, 2003d, 2003e, 2003f, 2003g). A summary of the current fishery and its management history based on these sources is provided below.

The Northeast skate complex is comprised of seven different skate species. These include the barndoor (*Dipturus laevis*), clearnose (*Raja eglanteria*), little (*Leucoraja erinacea*), rosette (*Leucoraja garmani*), smooth (*Malacoraja senta*), thorny (*Amblyraja radiata*), and winter (*Leucoraja ocellata*) skates. The seven species of skates are distributed along the coast of the northeast United States from near the tide-line to depths exceeding 700 meters (383 fathoms). Within the complex, the ranges of the individual species vary. In the Northeast Region, the center of distribution for the little and winter skates is Georges Bank (GB) and Southern New England (SNE). The barndoor skate is most common in the offshore Gulf of Maine (GOM), on GB, and in SNE along the shelf edge. The thorny and smooth skates are commonly found in the GOM while the clearnose and rosette skates have a more southern distribution, and are found primarily south of the Chesapeake Bight. Skates are not known to make large-scale migrations, but they do move seasonally in response to changes in water temperature, moving offshore in summer and early fall and returning inshore during winter and spring.

Skates are harvested for two very different commercial markets—one market supplies whole skates to be used as bait in the lobster fishery, and one market supplies skate wings for human consumption. The skate bait fishery is a directed fishery and is more traditional, involving vessels primarily from SNE ports that target little skates (>90% of landings) and, to a much lesser extent, juvenile winter skates (<10% of landings). The vessels supplying skates for the bait market tend to make dedicated trips targeting skates and land large quantities of skates per trip. The vessels involved in the skate bait fishery primarily use bottom otter trawl gear.

The skate wing fishery developed in the 1990s when skates were promoted as “underutilized species,” and fishermen shifted effort from groundfish and other fisheries to skates and spiny dogfish. The wing fishery is almost entirely an incidental catch fishery that involves vessels that also participate in the groundfish, monkfish, and/or scallop fisheries.

Most skates are caught using trawls, although gillnets are also used. During the period of 2000-2007, trawl landings accounted for approximately 65-86% of all skate landings (wings and bait combined), with gillnets accounting for the vast majority of the remainder of the landings. However, from 2008-2011, trawl landings contributed only 35% of the total skate wing landings, demonstrating the increasing use of gillnets to harvest skate wings. Gillnet landings are predominantly wings (97-98%), and as described above, are, for the most part, incidental to other targeted species, namely multispecies (e.g., cod, haddock, pollock, plaice, halibut, redfish, hake) and monkfish (NEFMC 2009a). Very small amounts of landings (<1%) are associated with hook and line gear and scallop dredges. As hook and line and dredge gear are seldom used in the fishery, their effects on ESA-listed species are discountable and, as a result, will not be analyzed further in this Opinion.

Commercial landings of skates have increased slowly since 1996, mainly in the wing fishery, while the prices for skate landings have markedly increased since 2001. As a result of better markets and regulations in other fisheries, vessels appear to be increasing the number of skates they are landing for wings. Although discards have declined considerably since 2001, they still represent nearly 37% of the total skate catch. Since skates are hard to identify by species, much of the landings and some of the observed discards are reported as unclassified skates (Table 3).

**Table 3** Commercial landings and total catch (landings plus dead discards) of skates from 2006-2011 (NMFS/NERO/APSD)

| Year | Landings (mt) | Total Catch (mt) |
|---|---|---|
| 2006 | 16,933 | 28,132 |
| 2007 | 20,086 | 34,562 |
| 2008 | 20,945 | 32,627 |
| 2009 | 20,738 | 30,308 |
| 2010 | 19,430 | 31,804 |
| 2011 | 16,586 | 29,086 |

The directed skate bait fishery is dominated by 20-30 Rhode Island vessels, while a smaller number of vessels from other SNE and northeast U.S. ports also participate in the fishery. The directed skate bait fishery operates throughout the year, peaking in the spring (with the increase in lobster fishing) and running until early winter. This fishery catches almost entirely skates, with little landings of other species. Most bait landings come from NMFS statistical areas 537 and 539, and, as noted above, go to ports in Rhode Island (Table 4). Although VTRs cannot be used to differentiate areas fished for directed bait versus wings, industry reports and information from Rhode Island suggest that almost all directed bait landings come from these two statistical areas.

**Table 4** Primary ports associated with the skate wing and bait fisheries in 2011 (NMFS/NERO/APSD)

| Top Bait Ports | 2011 Landings (million lb) | Top Wing Ports | 2011 Landings (million lb) |
|---|---|---|---|
| Point Judith, RI | 5.351 | Chatham, MA | 6.742 |
| New Bedford, MA | 2.437 | New Bedford, MA | 4.217 |
| Newport, RI | 1.766 | Point Judith, RI | 3.819 |
| New Jersey | 0.521 | Long Beach, NJ | 1.542 |
| Connecticut | 0.024 | Little Compton, RI | 1.470 |

The directed bait fishery occurs primarily in federal waters less than 40 fathoms deep from the Southern Massachusetts/Rhode Island/Connecticut/New York state waters boundary east to the waters south of Martha's Vineyard and Nantucket out to approximately 69° W longitude. Effort in state waters increases seasonally to accommodate the amplified effort in the spring to fall lobster fishery.

Skates are the preferred bait for the SNE inshore and offshore lobster pot fishermen, as skate meat is tough and holds up longer in the pot than other soft bait choices. Size drives the dockside price for bait skate, with "dinner plate" being the preferable size to be strung and placed inside lobster pots. Little and winter skates are rarely sorted prior to landing, as they are very similar. Documented skate landings increased during the 1990s (from 6,700 metric tons in 1989 to an average of around 11,400 metric tons annually from 1990 to 2003). Fishermen and state fisheries managers attribute the increase in skate landings in the 1990s to better reporting and documentation rather than a significant expansion of the skate fishery. The increase in Rhode Island skate fishery landings is coincident with the state's implementation of a comprehensive system to document commercial fishery landings data.

As of September 18, 2003 (the effective date of the Skate FMP), commercial fishermen must have a federal open access Skate Permit to possess or land skates in or from federal waters. Federally permitted vessels and vessels fishing in federal waters are prohibited from retaining, possessing, or landing barndoor and thorny skates throughout the Northeast Region. Additionally, these vessels are prohibited from retaining, possessing, or landing smooth skates from within the GOM Regulated Mesh Area. The other four skate species may be retained in accordance with the federal skate regulations found in 50 CFR Part 648, subpart O.

Although the management unit for the Skate FMP is broadly defined as U.S. waters in the western Atlantic from Maine through Cape Hatteras, NC, the fishery does not operate at all times and in all areas of the management unit. In U.S. Atlantic waters, peaks in landings occur in certain seasons and locations. These peaks may be influenced by management measures, market conditions, weather, spawning, and coastal migrations, among other factors.

The regulations implementing the Skate FMP require the NEFMC to monitor the status of the skates and the fishery on an annual basis. The regulations include the following: permit requirements for vessels possessing skates and dealers purchasing skates; reporting requirements; possession limits for skate wings and bait; an exemption from the wing possession limit for vessels fishing only for skates for the bait market; and prohibitions on the possession of smooth skates from or in the GOM, and barndoor and thorny skates throughout their range. The Skate FMP was developed, in part, to collect more complete and accurate information on the catch and disposition of skates in Northeast fisheries, particularly at the species level. Prior to the Skate FMP, all skate catch was categorized generally as “skate spp.” Stock assessments and efforts to manage fishing mortality have been hampered by a lack of species-specific catch information.
Even though skates are managed under a federal FMP, reported landings remain incomplete at the species level due to issues with species identification. Although some skates are caught by recreational fishermen, recreational landings of skates are negligible both in the context of all recreational fisheries (0.015% of all Atlantic coast recreational landings) and in the context of the overall skate fisheries (0.085% of all skate landings).

Skate fishery specifications for the 2012-2013 fishing years were implemented on May 1, 2012 (77 FR 25097). The specifications included an annual catch limit (ACL) for all skates combined of 50,435 metric tons, an annual catch target of 37,826 metric tons (75% of ACL), and allowable landing quotas for the wing fishery (14,338 mt) and bait fishery (7,223 mt). Possession limits in the wing fishery are 2,600 pounds wing weight in Season I (May 1–August 31) and 4,100 pounds wing weight in Season II (September 1–April 30) for vessels fishing on a NE Multispecies, Monkfish, or Scallop DAS. Vessels in the bait fishery, and carrying a Skate Bait Letter of Authorization, have a 25,000-pound whole weight possession limit. There is an incidental possession limit of 500 pounds of wings for vessels not fishing on a DAS.

### 3.7 Description of the Current Mackerel/Squid/Butterfish Fishery

The proposed action includes the continued operation of the MSB fishery managed under the Atlantic Mackerel, Squid and Butterfish FMP including measures implemented by Framework Adjustment 6, Framework Adjustment 7, and new specification and management measures for the 2013 fishing year, as discussed below.

Framework Adjustment 6, which became effective August 2012, adjusts the Council’s risk policy and is intended to prevent overfishing when no Overfishing Limit (OFL) or OFL proxy is available. Specifically, Framework Adjustment 6 defines the circumstances under which ABC can be increased if no OFL or OFL proxy is available, and eliminates the previous conflicting policies with a more clearly defined rule. Though Framework Adjustment 6 only modifies the Atlantic Mackerel/Squid/Butterfish FMP, it applies to all of the Council's managed species, including Atlantic mackerel, butterfish, Atlantic bluefish, spiny dogfish, summer flounder, scup, black sea bass, Atlantic surfclam, ocean quahog, and tilefish. The regulations for the risk policy reside in the Atlantic Mackerel/Squid/Butterfish FMP, but are a product of the Omnibus Amendment, which affected all of the plans for the above listed species.

Framework Adjustment 7, which became effective March 2013, changes the butterfish catch (discards and landings) cap into a butterfish discard (just discards) cap to account for the proposed directed butterfish fishery. There is no change to the total control of butterfish catch and the change is primarily an administrative adjustment to account for expected directed butterfish fishing in 2013.

Specifications and management measures for the 2013 fishing year were implemented January 16, 2013. The Atlantic mackerel quota is unchanged from 2012 and is being implemented for 3 years, from 2013 through 2015. The butterfish quota is being implemented for 2013 only, and is an increase of 1,698 mt over the 2012 quota (872 mt). The butterfish mortality cap is a 1,299-mt increase over the current 2012 cap level (3,165 mt). This action also proposes changes to butterfish possession limits and quota closure thresholds due to the proposed increase in the butterfish quota for 2013 and the potential for an increase in directed butterfish fishing. While some of the butterfish quota may be caught on other fishing trips, due to the increase in butterfish quota, there is likely to be some increase in directed butterfish fishing effort in 2013.

### 3.7.1 Description of the Atlantic Mackerel Fishery

The bulk of commercial Atlantic mackerel landings occur in the early part of the year from January-April (Clark 1998; Amendment 10 Draft EIS). During these months, the stock tends to be in shallower water and is more accessible to commercial harvest. An Atlantic mackerel trawl fishery also occurs in the GOM during the summer and fall months (May-December) (Clark 1998). Geographically, Atlantic mackerel harvest is widely distributed between Maine and North Carolina. Concentrations of catch occur on the continental shelf southeast of Long Island, NY and east of the Delmarva Peninsula.

The primary participants are generally larger vessels, averaging 112 feet, about 1700 horsepower with a crew of seven, which either freeze their catch on board or keep it in refrigerated seawater and process it on shore. Larger vessels ranging from 50 to 160 feet carry three to four fishermen on average, however, vessels that freeze and process fish at sea may carry 10 to 12 crewmen. These larger vessels run from 1-18 day trips, depending upon the vessel's capability to store catch and meet quota. Vessels that do not freeze and process at sea are known as "wet boats;" these vessels either ice their catch or store it in refrigerated sea water for up to seven days. Vessels that freeze at sea have the ability to make longer trips, averaging 12-14 days and extending as long as 18 days at sea.

The secondary participants are generally medium size vessels, averaging 72 feet, about 650 horsepower with a crew of four, who handle their catches in a variety of ways as there is great diversity of vessels among the smaller participants.

The status of the Atlantic mackerel was reassessed by the Transboundary Resource Assessment Committee (TRAC) in March 2010 (TRAC Report 2010/11). The TRAC status report indicated reduced productivity in the stock and a lack of older fish in both the survey and catch data. Though the status of the mackerel stock is still officially listed as "not overfished/overfishing not occurring," the TRAC assessment was not able to generate biomass reference points, and the stock status according to the most recent assessment is unknown.

Mackerel are taken with a variety of gears but mostly bottom otter trawl, single midwater trawls, and paired midwater trawls. Landings by gear type as recorded in the NMFS dealer weigh-out database 1982-2010 are displayed below in Table 5. Based on the NE Dealer Weigh-Out Database, the vast majority of commercial Atlantic mackerel landings are taken by trawl gear. Among trawl types, unspecified midwater otter trawls and paired midwater trawls have become increasingly important in recent years.

From 2002 to 2006, paired midwater trawls comprised 38% of commercial Atlantic mackerel landings, while unspecified midwater trawls also accounted for 40% of the landings, and bottom otter trawls comprised only 14% of the landings. In the last five years (2006-2010) the bottom otter trawl component of the fishery has increased slightly to reach roughly 23% of the overall landings. By comparison, from 1996 to 2000, paired midwater trawls landings comprised only 2% of the total commercial Atlantic mackerel landings, while unspecified midwater trawls accounted for 22% of the landings, and bottom otter trawls accounted for 71% of the landings. Since 2001, the great majority of mackerel have been landed by single and paired midwater trawls. Landings have varied by year, but paired trawls have taken the greatest quota of mackerel.

**Table 5** Landings by Gear, (NMFS Dealer Weigh-Out Data): Mackerel landings by gear type, total landings, quota, percent of quota and Initial Optimum Yield (IOY). IOY is a reduction of Allowable Biological Catch (ABC) that accounts for management uncertainty.

| Year | Bottom Otter Trawl | Single Midwater Trawl | Paired Midwater Trawl | Other | Total | Initial Optimum Yield IOY | Percent of IOY Landed |
|---|---|---|---|---|---|---|---|
| 1982 | 1,908 | . | 19 | 744 | 2,671 | | |
| 1983 | 890 | . | 410 | 1,342 | 2,642 | | |
| 1984 | 1,235 | 118 | 396 | 1,045 | 2,795 | | |
| 1985 | 1,481 | . | 249 | 905 | 2,635 | | |
| 1986 | 3,436 | . | 2 | 514 | 3,951 | | |
| 1987 | 3,690 | . | 0 | 649 | 4,339 | | |
| 1988 | 5,770 | . | 0 | 562 | 6,332 | | |
| 1989 | 7,655 | . | 0 | 589 | 8,245 | | |
| 1990 | 8,847 | . | 0 | 1,031 | 9,878 | | |
| 1991 | 15,514 | 564 | 223 | 285 | 16,585 | | |
| 1992 | 11,302 | . | 1 | 458 | 11,761 | | |
| 1993 | 3,762 | 479 | . | 412 | 4,653 | | |
| 1994 | 8,366 | 1 | . | 551 | 8,917 | | |
| 1995 | 7,920 | 50 | . | 499 | 8,468 | 100,000 | 8% |
| 1996 | 13,345 | 1,295 | . | 1,088 | 15,728 | 105,500 | 15% |
| 1997 | 13,927 | 628 | . | 847 | 15,403 | 90,000 | 17% |
| 1998 | 12,095 | 571 | 1,363 | 495 | 14,525 | 80,000 | 18% |
| 1999 | 11,181 | 99 | . | 752 | 12,031 | 75,000 | 16% |
| 2000 | 4,551 | 736 | . | 362 | 5,649 | 75,000 | 8% |
| 2001 | 584 | 11,396 | . | 360 | 12,340 | 85,000 | 15% |
| 2002 | 4,008 | 11,669 | 10,477 | 376 | 26,530 | 85,000 | 31% |
| 2003 | 5,291 | 17,212 | 11,572 | 222 | 34,298 | 175,000 | 20% |
| 2004 | 7,329 | 23,170 | 20,499 | 5,440 | 56,438 | 170,000 | 33% |
| 2005 | 5,437 | 15,635 | 18,894 | 2,242 | 42,209 | 115,000 | 37% |
| 2006 | 10,359 | 24,413 | 19,360 | 2,509 | 56,641 | 115,000 | 49% |
| 2007 | 2,097 | 14,715 | 8,080 | 655 | 25,547 | 115,000 | 22% |
| 2008 | 9,472 | 2,727 | 9,137 | 413 | 22,439 | 115,000 | 20% |
| 2009 | 6,758 | 9,318 | 5,670 | 890 | 22,634 | 115,000 | 20% |
| 2010 | 2,744 | 1,992 | 4,149 | 1,006 | 9,891 | 115,000 | 9% |

Atlantic mackerel are caught throughout the New England and Mid-Atlantic region but were generally concentrated off the coast of Delmarva through Rhode Island for the years 1998-2002. From 2003 to 2010 (the last year for which we have complete results), the southern areas have seen a reduction in landing activity while the northern states, particularly Massachusetts, have seen an increase. In the last four years, overall landings from all gear types have seen a sharp decline. It is not entirely clear why catches have not approached the quotas in recent years. A mix of factors may be involved, including market forces that affect fishing incentives (e.g. costs of inputs like fuel and prices fishermen can get for mackerel) and environmental forces that affect mackerel recruitment and abundance and/or availability in given locations. Fishermen have reported to the Council that they have been unable to find mackerel in sufficient quantity and density to harvest the quota, which supports the availability issue. For 2010, the top three states for mackerel landings (metric tons) were Massachusetts (MA) 56%, New Jersey (NJ) 22% and Rhode Island (RI) 21%. Of particular note, three of the top five commercial ports that land mackerel are in Massachusetts (2010).

The mackerel stock is the only stock in the Atlantic Mackerel, Squid, and Butterfish FMP that has a notable recreational component. Mackerel are seasonally important to the recreational fisheries of the Mid-Atlantic and New England regions. Recreational anglers catch mackerel in the Mid-Atlantic primarily during the spring migration, although this fishery has not been as robust in recent years. Historically, mackerel first appear off Virginia in March and gradually move northward. Christensen *et al.* (1979) found mackerel to be available to the recreational fishery from Delaware to New York for about three weeks (generally from early April to early May). The annual recreational catch of mackerel appears to be sensitive to changes in their migration and subsequent distribution pattern (Overholtz *et al.* 1989). In recent years, recreational mackerel harvest has varied from roughly 1,633 metric tons in 1997 to 530 metric tons in 2004, and an additional 10% of all mackerel caught (by number) were released. The highest landings occur from Massachusetts to Maine. Most mackerel are taken from boats.

### 3.7.2 Description of the Short-fin Offshore (*Illex*) Squid Fishery

The U.S. domestic fishery for *Illex s*quid, ranging from southern New England to Cape Hatteras, NC, reflects patterns in the seasonal distribution of *Illex* squid (*Illex illecebrosus*). Because *Illex* geographical range extends well beyond the U.S. EEZ, *Illex* are subject to exploitation in waters outside the U.S. jurisdiction. During the mid-1970s, a large directed fishery for *Illex* developed in the North Atlantic Fishery Organization (NAFO) subareas. *Illex* are harvested offshore (along or outside of the 100 meter isobath), mainly by small-mesh otter trawlers, when the squid are distributed in continental shelf and slope waters during the summer months (June-September) (Clark 1998). U.S. landings of *Illex* between 1982 and 2006 have fluctuated from 1,428 metric tons in 1983 to 26,097 metric tons in 2004. Landings for *Illex* peaked in 2004. Since 2004, landings have been down roughly 40%. Up to 2004 there was a relatively steady increase in landings that peaked in the mid-1990s and then generally declined. Two exceptional years since the mid-1990s peak were 1998 (23,568 metric tons) and 2004 (26,097 metric tons), resulting in closures of the directed fishery because the domestic quota was exceeded by 24% and 8.7%, respectively. The vast majority of U.S. commercial landings are taken by bottom otter trawls (see Table 6). The bulk of commercial landings for *Illex* occur between May-October.

The temporal patterns of the *Illex* fisheries in both U.S. and Canadian waters are determined primarily by the timing of the species' spawning migration to the continental shelf, although worldwide squid market conditions also influence the timing of the fishing season in the U.S. EEZ (NEFSC 2003). According to NEFSC (2003), the largest contribution to total *Illex* landings tends to occur along the continental shelf break in depths between 128 and 366 meters (70-200 fathoms). Although *Illex* are a ubiquitous bait item used in recreational fishing activities, these bait squid are a product of the commercial fishery and are, therefore, already accounted for in the recorded commercial fishery landings. There is no directed recreational fishery for *Illex* of any significance.

The *Illex* stock was most recently assessed at SARC 42 (2006). SARC 42 was publically available in 2006 and included data through 2004. It was not possible to evaluate current stock status because there were no reliable current estimates of stock biomass or fishing mortality rate. The short lifespan of *Illex* greatly complicates assessing the stock with the available survey and assessment resources. However, based on a number of qualitative analyses, it was determined that overfishing was not likely to have occurred during 1999-2002.

**Table 6** *Illex* landings by gear type, total landings, quota, percent of quota (Dealer Weigh-Out Data) and Initial Optimum Yield (IOY). IOY is a reduction of Allowable Biological Catch (ABC) that accounts for management uncertainty.

| YEAR | Bottom Otter Trawl | Other | TOTAL | Initial Optimum Yield IOY | Percent of IOY Landed |
|---|---|---|---|---|---|
| 1982 | 3,530 | 3 | 3,533 | | |
| 1983 | 1,413 | 16 | 1,428 | | |
| 1984 | 3,287 | 3 | 3,290 | | |
| 1985 | 2,447 | 0 | 2,447 | | |
| 1986 | 4,408 | 1 | 4,409 | | |
| 1987 | 6,468 | 494 | 6,962 | | |
| 1988 | 1,953 | 4 | 1,957 | | |
| 1989 | 6,801 | 0 | 6,801 | | |
| 1990 | 11,315 | 0 | 11,316 | | |
| 1991 | 11,906 | 2 | 11,908 | | |
| 1992 | 17,822 | 5 | 17,827 | | |
| 1993 | 18,012 | 0 | 18,012 | | |
| 1994 | 17,693 | 657 | 18,350 | | |
| 1995 | 13,970 | 6 | 13,976 | | |
| 1996 | 15,690 | 1,279 | 16,969 | | |

| | | | | | |
|---|---|---|---|---|---|
| 1997 | 13,004 | 352 | 13,356 | | |
| 1998 | 23,219 | 349 | 23,568 | 19,000 | 124% |
| 1999 | 7,309 | 80 | 7,389 | 19,000 | 39% |
| 2000 | 8,967 | 44 | 9,011 | 24,000 | 38% |
| 2001 | 4,009 | 0 | 4,009 | 24,000 | 17% |
| 2002 | 2,709 | 41 | 2,750 | 24,000 | 11% |
| 2003 | 6,111 | 280 | 6,391 | 24,000 | 27% |
| 2004 | 24,428 | 1,669 | 26,097 | 24,000 | 109% |
| 2005 | 7,955 | 4,057 | 12,011 | 24,000 | 50% |
| 2006 | 13,447 | 497 | 13,944 | 24,000 | 58% |
| 2007 | 7,948 | 1,074 | 9,022 | 24,000 | 38% |
| 2008 | 12,710 | 3,190 | 15,900 | 24,000 | 66% |
| 2009 | 17,804 | 614 | 18,418 | 24,000 | 77% |
| 2010 | 11,586 | 4,239 | 15,825 | 24,000 | 66% |

### 3.7.3 Description of the Longfin Inshore Squid Fishery

Based on a new proposed biomass reference point from the 2010 assessment (NEFSC 2011), the longfin inshore squid (longfin squid, or (*Doryteuthis (Amerigo) pealeii*) stock was not overfished in 2009, but overfishing status was not determined because no overfishing threshold was recommended. The 2010 longfin squid assessment (NEFSC 2011) found that the longfin squid stock appears to have successfully supported the range of observed catches (9,600 metric tons - 26,100 metric tons) during 1976-2009.

The U.S. domestic fishery for longfin squid occurs mainly in southern New England and Mid-Atlantic waters. Fishery patterns reflect longfin squid’s seasonal distribution, therefore most effort is directed offshore near the edge of the continental shelf during the fall and winter months (October-March) and inshore during the spring and summer months (April-September) (Clark 1998). Longfin squid are primarily harvested by bottom otter trawl gear (Table 7).

**Table 7** Longfin squid landings by gear type, total landings, quota, percent of quota (Dealer Weigh-Out Data) and Initial Optimum Yield (IOY). IOY is a reduction of Allowable Biological Catch (ABC) that accounts for management uncertainty**.**

| YEAR | Bottom Otter Trawl | Single Midwater Trawl | Dredge (for unknown species) | All others | Total | IOY | Percent of IOY Landed |
|---|---|---|---|---|---|---|---|
| 1982 | 2,445 | 0 | . | 79 | 2,524 | | |
| 1983 | 8,266 | . | . | 466 | 8,731 | | |
| 1984 | 6,648 | . | . | 509 | 7,158 | | |
| 1985 | 6,217 | . | . | 647 | 6,864 | | |
| 1986 | 10,867 | . | . | 646 | 11,512 | | |
| 1987 | 9,699 | . | . | 655 | 10,354 | | |
| 1988 | 16,811 | . | . | 1,751 | 18,562 | | |
| 1989 | 22,416 | . | . | 1,234 | 23,650 | | |
| 1990 | 14,354 | . | . | 599 | 14,954 | | |
| 1991 | 18,849 | 3 | . | 557 | 19,409 | | |
| 1992 | 17,914 | . | . | 263 | 18,177 | | |
| 1993 | 21,885 | . | . | 386 | 22,272 | | |
| 1994 | 22,404 | . | . | 159 | 22,563 | | |
| 1995 | 17,622 | . | . | 725 | 18,348 | | |
| 1996 | 11,720 | 440 | . | 254 | 12,414 | | |
| 1997 | 15,649 | 2 | . | 461 | 16,113 | | |
| 1998 | 18,962 | 2 | . | 159 | 19,123 | 21,000 | 91% |
| 1999 | 18,938 | 0 | . | 171 | 19,109 | 21,000 | 91% |
| 2000 | 17,198 | 23 | . | 259 | 17,480 | 13,000 | 134% |
| 2001 | 14,021 | 45 | . | 171 | 14,238 | 17,000 | 84% |
| 2002 | 16,508 | . | . | 198 | 16,707 | 17,000 | 98% |
| 2003 | 11,839 | . | . | 96 | 11,935 | 17,000 | 70% |
| 2004 | 12,874 | 493 | 364 | 1,834 | 15,566 | 17,000 | 92% |
| 2005 | 11,673 | 1,290 | 1,037 | 2,982 | 16,983 | 17,000 | 100% |
| 2006 | 12,577 | 333 | 892 | 2,105 | 15,907 | 17,000 | 94% |
| 2007 | 9,990 | 272 | 602 | 1,477 | 12,342 | 17,000 | 73% |
| 2008 | 9,503 | | 368 | 1,530 | 11,400 | 17,000 | 67% |
| 2009 | 7,857 | 88 | 192 | 1,171 | 9,306 | 19,000 | 49% |
| 2010 | 5,359 | 215 | | 1,028 | 6,855 | 18,667 | 37% |

Patterns of commercial harvest of longfin squid have complicated seasonal and annual distribution patterns (Macy and Brodziak 2001; Hatfield and Cadrin 2002). Depending on season and water temperatures, this species is distributed from relatively shallow nearshore areas, across the continental shelf, and on the upper continental slope, with the largest individuals in relatively deep water (Cadrin and Hatfield 1999). Commercial longfin squid landings generally peak in the spring and fall. Landings of longfin squid early in the year occur near the continental shelf break (102–183 meters [56-100 fathoms]; Hendrickson 2006), while summer and fall landings are harvested predominately near shore.

### 3.7.4 Description of the Butterfish Fishery

Beginning in 1963, vessels from Japan, Poland and the USSR began to exploit butterfish along the edge of the continental shelf during the late autumn through early spring. Reported foreign catches of butterfish increased from 750 metric tons in 1965 to 15,000 metric tons in 1969, and then to about 18,000 metric tons in 1973. With the advent of extended jurisdiction in U.S. waters, reported foreign landings declined sharply from 10,353 metric tons in 1976 to 1,326 metric tons in 1978. Foreign landings were slowly eliminated by 1987.

A peak in U.S. commercial butterfish landings (11,300 metric tons) occurred in 1984. Relatively high landings levels in the 1980s were attributed to heavy demand for butterfish in the Japanese market (NEFSC 2004). Demand from that market has since waned and landings averaged only 2,790 metric tons during 1990-1999. Since 2001, there has been minimal directed fishing, so landings have been very low, ranging from 437 to 872 metric tons during 2002-2010. Most landed butterfish are currently caught incidentally when other species, principally squid, are being targeted.

Of the 64,088 individual hauls monitored through the Northeast Fisheries Observer Program (NEFOP) from 2001 to 2010, only 36 hauls (~0.06 of 1%) indicated butterfish as the primary target species, yet butterfish were retained on 901 (~18%) of the observed trips. As such, it is difficult to characterize the trips that contribute to the majority of butterfish landings. Fisheries with substantial butterfish bycatch include the longfin squid, silver hake, mackerel, and mixed groundfish fisheries. Of these fisheries, the largest and most consistent bycatch occurs in the small-mesh squid fisheries (NEFSC 2010). Between 2001 and 2009, the longfin squid fishery was responsible for 68% of butterfish discards.

Atlantic butterfish (*Peprilus triacanthus*) undergo a northerly inshore migration during the summer months, a southerly offshore migration during the winter months, and are mainly caught as bycatch in the directed longfin squid and mackerel fisheries. Fishery observers suggest that a significant amount of Atlantic butterfish discarding occurs at sea. From 1997 to 2001, the bulk of the U.S. commercial butterfish landings occurred in January-March. More recently (2001-2010), landings have been spread throughout the year (likely due to lack of directed effort), with a slight peak recorded between May and August for 2010. Although low-level butterfish harvest is widespread, concentrations of landings come from southern New England shelf break areas near 40º N, as well as in and near Long Island Sound. In 2010, two ports reported more than 50% of all landings (Point

Judith, RI and Montauk, NY). All other landing ports reported landings of less than 10%. When compared to the other three species managed by this FMP, the actual fishery for butterfish is minimal. Seventy percent of reported landings came from bottom otter trawl gear. From 2002 to 2010, the mean annual butterfish landings have been very low (~480 metric tons).

The 49$^{th}$ Northeast Regional Stock Assessment Workshop (SAW 49) results, published in January 2010, provided updated estimates of butterfish fishing mortality and stock biomass. The current status of the butterfish stock is unknown because biomass reference points could not be determined in the SAW 49 assessment. Though the butterfish population appeared to be declining for some time leading up to the 2010 assessment, fishing mortality did not seem to be the major cause. Butterfish have a high natural mortality rate, and the current estimated fishing mortality rate (F = 0.02) is well below all candidate overfishing threshold reference points. The assessment report noted that predation is likely an important component of the butterfish natural mortality rate (currently assumed to be 0.8), but also noted that estimates of consumption of butterfish by predators appear to be very low. Since the 2010 assessment, a number of state and federal trawl surveys have indicated that butterfish abundance may be increasing. The MAFMC has recommended increases to the butterfish catch limits for the 2012 and 2013 fishing years.

*Summary*

The federal MSB fishery is primarily a mobile gear fishery using midwater (both single and paired) and bottom otter trawl gear. The list of allowable commercial gear types authorized under this FMP as listed in the Federal Register under the List of Fisheries (64 FR 4030) includes trawl, pelagic drift gillnet, pelagic longline, hook and line/hand line, purse seine, pot, trap, dredge, and bandit gear. Other gear types, such as pound nets, may be used in state water fisheries. All non-trawl MSB gear types permitted and allowed to fish in the fishery constitute a minor part of the total effort in the fishery, and make up less than 3-4% of effort in the overall fishery (Dealer Database).

Several types of gillnet gear may be used in the MSB fishery, possibly by vessels catching mackerel to use as bait in tuna or lobster fisheries. In the last 10 years, these fisheries have declined. The bait component of this fishery, in particular, has greatly declined and is almost non-existent. Vessels using bait gillnets to harvest MSB species are required to possess a permit and comply with mandatory reporting requirements. Thus, even a bait gillnet vessel that does not sell mackerel but uses it to catch other species, such as lobster or tuna, is required to obtain a MSB permit and comply with mandatory reporting.

The Mid-Atlantic Bottom Trawl Fishery has been defined as a Category II fishery in the 2011 List of Fisheries (76 FR 73912, November 29, 2011), the list that classifies US commercial fisheries by their level of incidental mortality/serious injury to marine mammals. The MMPA defines Category II fisheries as causing "occasional incidental mortality or serious injury." There are at least two distinct components to this fishery. One is the mixed groundfish bottom trawl fishery and is managed via several FMPs. The second major component is the MSB fishery. This component is managed by the federal MSB FMP (50 CFR Part 648.20 through 648.24). The *Illex* and longfin squid fisheries are managed by moratorium permits, gear and area restrictions, annual quotas, and trip limits. A tiered limited access permit system, which features different possession limits for the different permit categories, is currently being implemented for the Atlantic mackerel fishery. Overall catch for Atlantic mackerel is controlled by an annual quota.

Total effort, measured in trips, for the Mid-Atlantic Mid-Water Trawl Fishery (both paired and single mid-water trawl for all trips from Massachusetts south) from 2006 to 2011 was 394, 366, 238, 265, 162, and 159, respectively (NMFS). During the period 2006-2011, estimated observer coverage (% of trips) was 9%, 7%, 29%, 39%, 78%, and 74% respectively (average 39%). While the rate of coverage for mid-water trawl trips is much higher than that for the previous 10 years (maximum 12.6% coverage, average 8.2% coverage, from 1997-2006), it is important to note that much of the increased coverage is related to pre-trip notification and observer coverage requirements instituted to monitor haddock bycatch on Atlantic herring trips in Closed Area I, rather than as a result of specific increased coverage for the Atlantic mackerel fishery. Nonetheless, this increased observer coverage trend for mid-water trawl trips contributes to a clearer understanding of MSB fishery.

**Table 8** Bottom trawl total effort (trips) for Atlantic mackerel and squid in the Mid-Atlantic region (bottom trawl only), and total effort for the Illex squid and longfin squid fisheries.

| | Mackerel | Illex | Longfin |
|---|---|---|---|
| 1997 | 373 | | |
| 1998 | 278 | 412 | 1048 |
| 1999 | 262 | 141 | 495 |
| 2000 | 102 | 108 | 529 |
| 2001 | 175 | 51 | 413 |
| 2002 | 310 | 39 | 3585 |
| 2003 | 238 | 103 | 1848 |
| 2004 | 231 | 445 | 1124 |
| 2005 | 0 | 181 | 1845 |
| 2006 | 117 | 159 | 3058 |

## 3.8 Description of the Current Summer Flounder/Scup/Black Sea Bass Fishery

The current management measures for the summer flounder, scup, and black sea bass fishery, the history of the fishery, and the general distribution and habitat preferences of the three species are described in the following documents referenced in the literature cited: NEFSC (2002, 2006, 2008), Shepherd (2006, 2009), Terceiro (2006a, 2006b, 2011a, 2011b), and MAFMC (2010, 2011a, 2011b). Additional information on the distribution and habitat characteristics of summer flounder, scup, and black sea bass can be found in the EFH source documents for the species (Packer *et al.* 1999; Steimle *et al.* 1999; Drohan *et al.* 2007). A summary of the current fishery and its management history based on these sources is provided below.

Summer flounder, scup, and black sea bass are managed by both the MAFMC and the ASMFC under a joint FMP. These species are managed under a single FMP because these species occupy similar habitat and are often caught at the same time. They are present in offshore waters of the U.S. Atlantic Ocean throughout the winter and migrate into and occupy inshore waters throughout the summer. Access to the commercial sector of each fishery is limited by moratorium permits. Summer flounder is projected to have exceeded the rebuilding threshold; however, a formal stock assessment update is currently being conducted by the NEFSC to confirm that the stock is indeed rebuilt. The most recently published stock assessment update indicated that the stock is neither overfished nor subject to overfishing (MAFMC 2011). Scup and black sea bass stocks are recently rebuilt and were not listed as overfished or subject to overfishing in the most recent stock assessment updates in support of the 2011 specifications (Shepherd 2009; Terceiro 2010).

Although managed under one FMP, permits for summer flounder, scup, and black sea bass are issued separately based on having met that fishery's limited access eligibility requirements. Each of these three commercial fisheries have vessels permitted as moratorium (or limited access) and open access charter/party or both. Of the vessels with at least one of these permits, 1,248 held only moratorium permits for summer flounder, scup, or black sea bass, with 563 active, while 889 held charter/party permits with 341 active (NMFS Permit and VTR databases, 2011). The largest number of commercial summer flounder, scup, and black sea bass permit holders are held by Massachusetts vessels, followed closely by New Jersey and New York, then Rhode Island and North Carolina. In terms of vessel size, the largest moratorium vessels within the management unit are found in Virginia, followed by Massachusetts, Connecticut, and North Carolina. The fewest number of permits and smallest vessels used in the fishery are held by Delaware permit holders.

Commercial landings by state have varied over recent years (2004-2009). For combined FMP landings, North Carolina (20.6%) and New Jersey (20.5%) had the highest percentage of landings from 2004 to 2009, with Rhode Island, New York, and Virginia close behind (18.5%, 16.5 %, and 15.5%, respectively). For summer flounder, North Carolina had the highest landings during the same time period, followed by Virginia and New Jersey. New York led in scup landings from 2004 to 2009, followed by Rhode Island and New Jersey. These three states accounted for almost 86% of the coastwide scup landings during that period. The most recent records (2004-2009) indicate that North Carolina (26%) had the highest commercial black sea bass landings, followed by New Jersey (23%). However, the historical distribution of commercial black sea bass landings by state has fluctuated since 1950. Virginia has generally had the highest black sea bass landings accounting for

42% of the total landings from Maine through North Carolina from 1950-2002, followed by New Jersey.

The primary gear types used in the summer flounder, scup, and black sea bass fisheries are mobile trawl gear, pots and traps, gillnets, pound nets, and handlines. Traditionally, the two main gear types in the black sea bass fishery are otter trawls (40%) and pot/trap gear (45%), which have accounted for about 85% of the coastwide landings from 1990 to 2008. Bottom trawling is the predominant gear type used in the summer flounder and scup fisheries, accounting for 93% and 75.3% of the fisheries landings, respectively. The other predominant gear is the shallow floating trap, which accounts for about 10% of the landings. Other gears that caught more than 1% of the landings include mid-water paired trawl, fish pot/traps, and handlines. Trap/pot gear accounts for a much smaller percentage of the overall scup effort than is found in the black sea bass fishery.

The summer flounder, scup, and black sea bass stocks are managed collaboratively between the MAFMC and NMFS, who manage them in federal waters (3-200 nautical miles offshore), and the individual states from Maine to North Carolina through the ASMFC, whose jurisdiction is for state waters (0-3 nautical miles offshore).

NMFS implemented ACLs and AMs for the summer flounder, scup, and black sea bass fisheries through the Mid-Atlantic Annual Catch Limit/Accountability Measure (ACL/AM) Omnibus Amendment. Recommendations for acceptable biological catch (ABC) are provided on an annual basis by the MAFMC's SSC, which sets ABC based on scientific uncertainty associated with catch levels that would result in overfishing the stock. For consistency with the requirements of the reauthorized MSA, the ACLs for summer flounder, scup, and black sea bass cannot be greater than the ABCs. Annual Catch Targets (ACTs) are set equal to or lower than the ACLs to account for management uncertainty in the fisheries before sector-specific landing limits (i.e., quotas) are derived for the commercial and recreational sectors. The commercial quota for summer flounder is managed on a state-by-state basis. For scup, the commercial quota is divided into three harvest periods. Federal waters are managed on a coastwide basis for each quota period and on a state-by-state basis by the ASMFC during the summer quota period and coastwide during the winter quota periods. The black sea bass commercial quota is managed on a coastwide basis in federal waters and on a state-by-state basis by the ASMFC. It should be noted that this patchwork of state and federal management makes projecting how effort will be distributed challenging; for example, when quotas go up, states may liberalize trip limits for the three species, allowing for more efficient operations at similar trip and tow length.

Quota specifications for the three species regulated under the FMP are generally set on an annual basis, but may be proposed for a three-year period. The most recent peer-reviewed assessment of the species found that based on the biological reference points, the stocks of each species are not overfished and overfishing is not occurring (MAFMC 2011). For 2012, NMFS set the commercial quota for summer flounder at 13.14 million pounds, for scup at 27.91 million pounds, and for black sea bass at 1.71 million pounds. Additionally, in 2012 NMFS increased the current scup commercial Winter I period possession limit from 30,000 to 50,000 pounds. Under the Mid-Atlantic Research Set Aside (RSA) program, 3% of each species' Total Allowable Landings (TAL) will continue to be set aside for research and improved data collection. For the recreational fisheries, the 2012 harvest limits were set at 8.76 million pounds for summer flounder, 8.45 million pounds for scup, and 1.32 million pounds for black sea bass.

NMFS implemented changes to management measures for the 2012 recreational fisheries (May 8, 2012, 77 FR 30427). Summer flounder conservation equivalency measures allow states to implement their own state-specific measures, as long as they produce the same conservation result. The precautionary default measures for summer flounder include a 20-inch total length (TL) minimum fish size, a two fish per person possession limit, and open season from May 1 through September 30. For scup, the measures included a 10.5-inch TL minimum fish size, a 20 fish per person possession limit, and open season of January 1 to December 31. For black sea bass, the measures included a 12.5-inch TL minimum fish size, 15 fish per person possession limit from January 1 to February 29, and a 12.5-inch TL minimum fish size, 25 fish per person possession limit from May 19 to October 14 and November 1 to December 31. All other management measures in the fishery remain the same (77 FR 30427).

A regulatory action to implement specifications and management measures for the 2013 fishing year went into effect January 1, 2013. The measures would also implement 2014 commercial fishing quotas and recreational harvest limits for summer flounder and scup. This action would slightly decrease the quotas for summer flounder and scup, and black sea bass quotas are proposed to remain nearly status quo for the commercial and recreational fisheries. For 2013, NMFS set the commercial quota for summer flounder at 11.45 million pounds, for scup at 23.52 million pounds, and for black sea bass at 1.78 million pounds. For the recreational fisheries, the 2013 harvest limits were set at 7.62 million pounds for summer flounder, 7.56 million pounds for scup, and 1.84 million pounds for black sea bass. For 2014, NMFS set the commercial quota for summer flounder at 11.39 million pounds and 21.94 million pounds for scup. For the recreational fisheries, the 2013 harvest limits were set at 7.6 million pounds for summer flounder and 7.03 million pounds for scup. Under the Mid-Atlantic Research Set Aside (RSA) program, 3% of each species' Total Allowable Landings (TAL) will continue to be set aside for research and improved data collection. All other management measures in the fishery remain the same.

### 3.9 Summary by Gear Type

An FMP is the operational unit used for managing a fishery (or collection of fisheries) that targets the species specifically addressed in the FMP. While the FMP works well as the unit for planning and implementing fishing regulations, for fisheries that overlap in time and space it is often not the most efficient or appropriate unit for monitoring incidental bycatch occurring in a fishery. Fishing activity under the authority of many FMPs often occurs simultaneously and on the same vessel, as seen with the diversity of catch in individual gillnet hauls, for example, and landings from fishing trips that include many species managed under multiple FMPs and authorized through multiple permits (or allowed per incidental bycatch limits). For example, commercial fishing vessels operating out of New England ports that use gillnets often target, and catch, monkfish, skates, and some groundfish species. Even though monkfish, skates, and groundfish fishing regulations are implemented under three separate FMPs, in many cases the same vessels are catching and landing each of these species, often in the same net.

Fisheries of the northeast and Mid-Atlantic regions use diverse gear types, yet none of them are selective enough to catch only targeted species. Because of the variations in how fishing effort among FMPs is carried out (and variations in the resulting protected resources bycatch data products), a goal for this assessment is to analyze impacts of fishery effort by gear type and effects to protected resources listed under the ESA through an analysis of gear types associated with bycatch records. Table 9 identifies the gears likely used for landing fish within the FMPs analyzed in this Opinion. While the listed associations are based on information presented in the Standardized Bycatch Reporting Methodology (2007) and the U.S. National Bycatch Report (NMFS 2011c), it is intended to provide a general overview of associations.

**Table 9** Gears Likely to be Used for Landing Fish Within the Listed FMPs

| FMP | Gillnet | | | Bottom Otter Trawl | | Midwater Trawl | Pot / Trap | Bottom Longline | Hook and Line |
|---|---|---|---|---|---|---|---|---|---|
| | <5.5" | >=5.5" < 8" | >=8" | <5.5" | >=5.5" | | | | |
| NE Multispecies | | x | x | x | x | | | x | x |
| Monkfish | | x | x | | x | | x | x | |
| Spiny Dogfish | x | x | | x | x | | | x | |
| Bluefish | x | x | | | x | | | | x |
| Skate Complex | x | x | x | x | x | | | | |
| Mackerel, Squid, Butterfish | | | | x | x | x | | | |
| Summer Flounder, Scup, Black Sea Bass | | | | | x | | x | | x |

## 3.10 Exempted, Education, and Research Fishing Permits

Regulations at 50 CFR 600.745 allow the Northeast Regional Administrator to authorize the targeting or incidental harvest of species managed under an FMP or fishing activities that would otherwise be prohibited for scientific research, limited testing, public display, data collection, exploratory, health and safety, environmental cleanup, hazardous waste removal purposes, or for educational activities. Every year, NMFS NERO may issue a small number of exempted fishing permits (EFPs) and/or exempted educational activity authorizations (EEAAs) exempting the collection of a limited number of species from Northeast Federal waters from regulations implementing the appropriate FMP. Table 10 shows the number of EFPs and EEAAs for each fishery issued by NERO from 2007 to 2011. These EFPs and EEAAs involve fishing by commercial or research vessels that use similar or identical fishing methods as the seven fisheries that are the subject of this Opinion. The only differences involved (a) the use of modified gear, which was not authorized under the specific FMP at the time, or (b) requests for additional DAS or trips to closed areas beyond what the annual specifications for the fishery allowed.

For the total of 51 EFPs and 10 EEAAs examined between 2007 and 2011, we were able to conclude that in all cases, the types and rates of interactions with listed species from the EFP and EEAA activities would be similar to those analyzed in their respective Opinions. Given our past experience with and knowledge of the usual applicants (and when and where they fish), we expect that future EFPs and/or EEAAs would propose fishing types and associated fishing effort similar to previous EFPs/EEAAs and, therefore, not introduce a significant increase in effort levels for the seven fisheries considered in this Opinion. For example, issuance of an EFP to an active commercial vessel that is similar to the ones described above likely does not add additional effects compared to those that would otherwise accrue from the vessel's normal commercial activities. Similarly, issuance of an EFP or EEAA to a vessel to conduct a minimal number of tows/trips with trawl or gillnet gear likely would not add sufficient fishing effort to produce a detectable change in the overall amount of fishing effort in a given year. Therefore, we consider the future issuance of most EFPs and EEAAs by NMFS NERO to be within the scope of this Opinion. If an EFP or EEAA is proposed which modifies this agency action in a manner that causes an effect to listed species or critical habitat not considered in this Opinion (i.e., is beyond the scope of the fishery activity considered), then additional section 7 consultation would be necessary.

**Table 10** The number of EFPs and EEAAs issued by NERO (2007-2011).

| FMP | EFPs | EEAAs |
|---|---|---|
| NE Multispecies | 18 | 10 |
| Summer flounder/Scup/Black sea bass | 15 | 0 |
| Spiny dogfish | 4 | 0 |
| Squid/mackerel/butterfish | 1 | 0 |
| Bluefish | 2 | 0 |
| Skate | 2 | 0 |

| | | |
|---|---|---|
| Monkfish | 9 | 0 |
| **Total** | **51** | **10** |

Amendment 2 to the Monkfish FMP established the Monkfish RSA Program, which sets aside 500 monkfish DAS annually from the total number of monkfish DAS allocated to limited access monkfish vessels, to address monkfish research priorities identified by the Councils. Projects funded under an RSA DAS award must enhance understanding of the monkfish fishery resource or contribute to the body of information used in management decisions, including reducing bycatch of and interactions with protected species. From 2006 through 2011, 16 research projects were supported through Monkfish RSA allocations.

In 2001, Framework Adjustment 1 to the Summer Flounder, Scup, and Black Sea Bass FMP, Atlantic Mackerel, Squid, and Butterfish FMP, Bluefish FMP, and Tilefish FMP established a procedure through which RSA quotas are set aside as part of the Council's annual quota-setting process. The set-asides may be 0%-3% of each species' TAL. Projects must enhance understanding of the fishery resource and contribute to the body of information used in management decisions. From 2002 through 2011, 34 research projects were supported through Mid-Atlantic RSA allocations. All of these were consistent with the biological opinions issued for the fisheries and did not trigger reinitiation; we would therefore expect that future RSA would also be covered by the Opinions. However, as is the case with EFPs and EEAAs, if we determine that the magnitude and/or distribution of effort or the types of gears used in an RSA project are not within the scope of this Opinion, additional section 7 consultation would be necessary.

Amendment 3 to the Spiny Dogfish FMP includes a provision to establish an RSA provision of up to 3% of the annual total allowable landings (TAL). The Council proposed this provision for the 2013-2015 management measures, which were implemented May 1, 2013 (78 FR 15,674 March 12, 2013).

### 3.11 Fisheries Observer Programs

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

### 3.12 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 seven fisheries operate, broadly defined as all U.S. EEZ waters from Maine through Key West, FL and the adjoining state waters that are affected through the regulation of activities of federal permit holders fishing in those waters. The direct and indirect effects of the seven fisheries on ESA-listed species in the action area have been summarized as impacts resulting from: (1) entanglement, capture, or hooking of these species in fishing gear, (2) the operation of vessels in the fisheries, and (3) changes to these species' habitats and prey as a result of bottom trawl and gillnet gear used in the fisheries.

## 4.0 Status of the Species

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

| **Common name** | **Scientific name** | **ESA Status** |
|---|---|---|
| North Atlantic right whale | *Eubalaena glacialis* | Endangered |
| Humpback whale | *Megaptera novaengliae* | Endangered |
| Fin whale | *Balaenoptera physalus* | Endangered |
| Sei whale | *Balaenoptera borealis* | Endangered |
| Loggerhead sea turtle - NWA DPS[10] | *Caretta caretta* | Threatened |
| Leatherback sea turtle | *Dermochelys coriacea* | Endangered |
| Kemp's ridley sea turtle | *Lepidochelys kempii* | Endangered |

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

| | | |
|---|---|---|
| Green sea turtle | *Chelonia mydas* | Endangered[11] |
| Atlantic sturgeon | *Acipenser oxyrinchus oxyrinchus* | |
| Gulf of Maine (GOM) DPS | | Threatened |
| New York Bight (NYB) DPS | | Endangered |
| Chesapeake Bay (CB) DPS | | Endangered |
| Carolina DPS | | Endangered |
| South Atlantic (SA) DPS | | Endangered |
| Atlantic Salmon - GOM DPS | *Salmo salar* | Endangered |

[11] 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.

## 4.1 Species Not Likely to Be Adversely Affected

### 4.1.1 Hawksbill Sea Turtles

Hawksbill sea turtles are uncommon in the northern waters of the continental United States, but are widely distributed throughout the Caribbean Sea, off the coasts of Florida and Texas in the continental U.S., in the Greater and Lesser Antilles, and along the mainland of Central America south to Brazil (Lund 1985; Plotkin and Amos 1988; Amos 1989; Groombridge and Luxmoore 1989; Plotkin and Amos 1990; NMFS and USFWS 1998b; Meylan and Donnelly 1999). Hawksbills prefer coral reefs, such as those found in the Caribbean and Central America. Hawksbills feed primarily on a wide variety of sponges but also consume bryozoans, coelenterates, and mollusks. The Culebra Archipelago of Puerto Rico contains especially important foraging habitat for hawksbills. Nesting areas in the western North Atlantic include Puerto Rico and the Virgin Islands. There are accounts of hawksbills in South Florida and individuals have been sighted along the East Coast as far north as Massachusetts, although sightings north of Florida are rare. Hawksbills have been found stranded as far north as Cape Cod, Massachusetts; however, many of these strandings were observed after hurricanes or offshore storms. Of the seven fisheries, six do not occur in waters that are typically used by hawksbill sea turtles. Only the bluefish fishery may occur in waters typically used by hawksbill sea turtles. However, 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 bluefish fishery will adversely affect hawksbill sea turtles.

### 4.1.2 Shortnose Sturgeon

Shortnose sturgeon are primarily benthic fish that mainly occupy the deep channel sections of large rivers. They can be found in rivers along the western Atlantic coast from St. Johns River, FL (possibly extirpated from this system), to the Saint John River in New Brunswick, Canada. The species is anadromous in the southern portion of its range (*i.e.*, south of Chesapeake Bay), while some northern populations are amphidromous (NMFS 1998a). New tracking data indicate that shortnose sturgeon are capable of making coastal migrations, and fish have been tracked between several Maine rivers and down to the Merrimack River in Massachusetts. However, even in the Northeast where these coastal migrations have been documented, shortnose sturgeon do not appear to spend significant time in the marine environment. Since the seven fisheries do not operate in or near the rivers where concentrations of shortnose sturgeon predominantly are found and their time in the marine environment is very limited, it is highly unlikely that the fisheries will affect shortnose sturgeon.

### 4.1.3 Smalltooth Sawfish DPS

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, 2003d, 2009d). With the exception of the bluefish fishery, none of the seven fisheries overlap with smalltooth sawfish and therefore are not expected to have any interaction with smalltooth sawfish. The bluefish fishery is the only fishery to extend south past North Carolina but likely does not extend west or inshore of Key West, Florida, and the likelihood of the fishery overlapping with the smalltooth sawfish DPS is discountable. In the unlikely event that the bluefish 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 2009e), 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.

### 4.1.4 Corals

Acroporid (*i.e.*, elkhorn and staghorn) corals require relatively clear, well circulated water. Typical water temperatures in which these species occur range from 21º-29ºC, 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, FL. Elkhorn corals commonly grow in turbulent shallow water on the seaward face of reefs in waters ranging from 1-5 meters in depth, but have been found to 30 meters. Staghorn corals commonly grow in more protected, deeper waters ranging from 5-20 meters in depth and have been found in rare instances to 60 meters. 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 used 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 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. This section also 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.*, make any adverse effects on these species from the proposed action extremely unlikely to occur. Based on this information, effects of the seven fisheries on ESA-listed *Acropora* corals and their designated critical habitats are discountable.

#### 4.1.5 Johnson's Sea Grass

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 of coastline in southeastern Florida between Sebastian Inlet and north Biscayne Bay (NMFS 2002c). The bluefish fishery is the only fishery operating in southeastern Florida. 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.

#### 4.1.6 Sperm Whale

Sperm whales regularly occur in waters of the U.S. EEZ, but primarily are found on the continental shelf edge, over the continental slope, and into mid-ocean regions (Waring *et al.* 2007). In contrast, the seven fisheries operate in continental shelf waters. The average depth of sperm whale sightings observed during the CeTAP surveys was 1,792 meters (CeTAP 1982). Female sperm whales and young males almost always inhabit waters deeper than 1000 meters and at latitudes less than 40° N (Whitehead 2002). Sperm whales feed on large organisms that inhabit the deep ocean regions (Whitehead 2002). Calving for the species occurs in low latitude waters outside of the action area. Given that sperm whales are unlikely to occur in areas (based on water depth) where the fisheries operate, and given that the operation of the fisheries 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 seven fisheries is not likely to adversely affect sperm whales.

#### 4.1.7 Blue Whale

Blue whales do not regularly occur in waters of the U.S. EEZ (Waring *et al.* 2010). 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 multispecies fishery operates. Blue whales feed on euphausiids (krill) (Sears 2002) which are too small to be captured in fishing gear used in the seven fisheries. Given that the species is unlikely to occur in areas where the fisheries operate, and given that the operation of the fisheries 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 fisheries is not likely to adversely affect blue whales.

#### 4.1.8 Right Whale Critical Habitat

We have determined that the action being considered in this Opinion is not likely to adversely modify or destroy critical habitat that was designated for northern right whales in 1994.[12] This determination is based on the action's effects on the conservation value of the designated habitat. 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's Cape Cod Bay, Great South Channel, and in nearshore waters off Georgia and Florida (50 CFR 226.203). Cape Cod Bay and Great South Channel, which are located within the action area, were designated as critical habitat for northern 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 seven fisheries will not affect the availability of copepods for foraging right whales because copepods are too small to be captured in fishing gear.

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. Of the seven fisheries, only the bluefish fishery may overlap with the winter calving and nursery grounds. The environmental features

[12] The North Atlantic right whale and the North Pacific right whale were recognized as distinct species under the ESA in 2008 (73 FR 12024, March 6, 2008).

that have been correlated with the distribution of right whales in these waters include preferred water depths and water temperature (Keller *et al*. 2012). 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 proposed action 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.

#### 4.1.9 Atlantic Salmon Critical Habitat

We have determined that the action being considered in this Opinion is not likely to adversely modify or destroy critical habitat that was designated for the GOM DPS of Atlantic salmon on June 19, 2009 (74 FR 29300) and revised on August 10, 2009 to exclude trust and fee holdings of the Penobscot Indian Nation and a table was corrected (74 FR 39003; August 10, 2009). Because there is no Atlantic salmon critical habitat in the marine environment where the seven fisheries occur, it will not be considered further in this Opinion.

### 4.2 Status of Large Whales

All of the cetacean species considered in this Opinion were once the subject of commercial whaling, which likely caused their initial decline. Commercial whaling for right whales along the U.S. Atlantic coast peaked in the 18th century, but right whales continued to be taken opportunistically along the coast and in other areas of the North Atlantic into the early 20th century (Kenney 2002). Worldwide, humpback whales were often the first species to be targeted and frequently hunted to commercial extinction (Clapham *et al.* 1999), meaning that their numbers had been reduced so low by commercial exploitation that it was no longer profitable to target the species. Wide-scale exploitation of the more offshore fin whale occurred later with the introduction of steam-powered vessels and harpoon gun technology (Perry *et al.* 1999). 1999). Fin whales were given total protection in the North Atlantic in 1987, with the exception of an aboriginal subsistence whaling hunt for Greenland (Gambell 1993, Caulfield 1993). Sei whales became the target of modern commercial whalers in the late 19th and early 20th centuries after populations of other whales, including right, humpback, fin, and blue, had already been depleted. The species continued to be exploited in Iceland until 1986, even though measures to stop whaling of sei whales had been enacted in the 1970s (Perry *et al.* 1999). 1999). However, Iceland has increased its whaling activities in recent years and reported a catch of 136 whales in the 1988/89 and 1989/90 seasons (Perry *et al.* 1999), seven in 2006/07, and 273 in 2009/2010. In 2011 and 2012, Iceland temporarily suspended commercial whaling for fin whales due to decreased demand from Japan, but is expected to resume in 2013. Today, the greatest known threats to these cetaceans are ship strikes and gear interactions, although the number of each species affected by these activities does vary.

Information on the range-wide status of each species as it is listed under the ESA is included here to provide the reader with information on the status of each species. Additional background information on the range-wide status of these species can be found in a number of published documents, including recovery plans (NMFS 1991a, b; 2005a), the Marine Mammal Stock Assessment Reports (SAR) (*e.g*., Waring *et al.* 2011), status reviews (*e.g*., Conant *et al.* 2009), and other publications (*e.g.* Clapham *et al.* 1999; Perry *et al.* 1999; Best *et al.* 2001).

#### 4.2.1 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 southern and northern 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. Originally called the "northern right whale," it was listed as endangered under the Endangered Species Conservation Act, the precursor to the ESA in June 1970. The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

In December 2006, NMFS completed a comprehensive review of the status of right whales in the North Atlantic and North Pacific Oceans. Based on the findings from the status review, NMFS concluded that right whales in the Northern Hemisphere exist as two species: North Atlantic right whale (*Eubalaena glacialis*) and 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 migrated along the coast from northern Europe to northwest Africa. The current distribution and migration patterns of the eastern North Atlantic right whale population, if extant, are unknown. Sighting surveys from the eastern Atlantic Ocean suggest that right whales present in this region are rare (Best *et al.*, 2001) and it is unclear whether a viable population in the eastern North Atlantic still exists (Brown 1986, NMFS 1991a). Photo-identification work has shown that some of the whales observed in the eastern Atlantic were previously identified as western Atlantic right whales (Kenney 2002).

This Opinion will focus on the western North Atlantic right whale (*Eubalaena glacialis*), which occurs in the action area.

*Habitat and Distribution*
Western North Atlantic right whales generally occur from the southeast U.S. to Canada (*e.g.*, Bay of Fundy and Scotian Shelf) (Kenney 2002; Waring *et al.* 2013). Like other right whale species, they follow an annual pattern of migration between low latitude winter calving grounds and high latitude summer foraging grounds (Perry *et al.* 1999; Kenney 2002).

The distribution of right whales seems linked to the distribution of their principal zooplankton prey, calanoid copepods (Winn *et al.* 1986; NMFS 2005a; Baumgartner and Mate 2005; Waring *et al.* 2012). Right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990; Schevill *et al.* 1986; Watkins and Schevill 1982) and in the Great South Channel in May and June (Kenney *et al.* 1986; Payne *et al.* 1990; Kenney *et al.* 1995; Kenney 2001) where they have been observed feeding predominantly on copepods of the genera *Calanus* and *Pseudocalanus* (Baumgartner and Mate 2005; Waring *et al.* 2011). Right whales also frequent Stellwagen Bank and Jeffreys 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 note 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, NC. In the North Atlantic, it appears that not all reproductively active females return to the calving grounds each year (Kraus *et al.* 1986; Payne 1986). Patrician *et al.* (2009) analyzed photographs of a right whale calf sighted in the Great South Channel in June 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 off 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, 2009, 2010, and 2011, right whales were sighted on Jeffreys and Cashes Ledges, Stellwagen Bank, and Jordan Basin during December to February (Khan *et al.* 2009, 2010, 2011, 2012). 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.* 2012). On multiple days in December 2008, congregations of more than 40 individual right whales were observed in the Jordan Basin area of the Gulf of Maine, leading researchers to believe this may be a wintering ground (NOAA

2008). Telemetry data have shown lengthy and somewhat distant excursions into deep water off the continental shelf (Mate *et al.* 1997) as well as extensive movements over the continental shelf during the summer foraging period (Mate *et al.* 1992; Mate *et al.* 1997; Bowman 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) is one of only two sightings in the 20$^{th}$ 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 southeastern United States. The frequency with which right whales occur in offshore waters in the southeastern United States remains unclear (Waring *et al.* 2012).

*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 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 much greater than this estimate (Best *et al.* 2001). Based on a census of individual whales using photo-identification techniques and an assumption of mortality for those whales not seen in seven years, a total of 299 right whales was estimated in 1998 (Kraus *et al.* 2001), and a review of the photo-ID recapture database on October 21, 2011 indicated that 425 individually recognized whales were known to be alive during 2009 (Waring *et al.* 2013). Whales catalogued by this date included 20 of the 39 calves born during that year. Adding the 19 calves not yet catalogued brings the minimum number alive in 2009 to 444. This number represents a minimum population size. The minimum number alive population index for the years 1990-2009 suggests a positive and slowly accelerating trend in population size. These data reveal a significant increase in the number of catalogued whales with a geometric mean growth rate for the period of 2.6% (Waring *et al.* 2013).

A total of 316 right whale calves were born from 1993 to 2010 (Waring *et al.* 2012). The mean calf production for this 18-year period is estimated to be 17.5/year (Waring *et al.* 2012). Calving numbers have been variable, with large differences among years, including a second largest calving season in 2000/2001 with 31 right whale births (Waring *et al.* 2012). The three calving years (97/98; 98/99; 99/00) prior to this record year provided low recruitment levels with only 11 calves born. The last ten calving seasons (2000-2010) have been remarkably better with 31, 21, 19, 17, 28, 19, 23, 23, 39, and 19 births, respectively (Waring *et al.* 2012). However, the western North Atlantic stock has also continued to experience losses of calves, juveniles, and adults.

As is the case with other mammalian species, there is an interest in monitoring the number of females in this western North Atlantic right whale population since their numbers will affect the population trend (whether declining, increasing or stable). Kraus *et al.* (2007) reported that, as of 2005, 92 reproductively-active females had been identified, and Schick *et al.* (2009) estimated 97 breeding females. From 1983 to 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). By 2005, 16 right whales had produced at least six calves each, and four cows had at least seven calves. Two of these cows were at an age that indicated a reproductive life span of at least 31 years (Kraus *et al.* 2007). As described above, the 2000/2001-2006/2007 calving seasons had relatively high calf production and have included several first time mothers (*e.g.*, eight new mothers in 2000/2001). However, over the same time period, there have been continued losses to the western North Atlantic right whale population, including the death of mature females, as a result of anthropogenic mortality (like that described in Henry *et al.* 2011, below). Of the 12 serious injuries and mortalities in 2005-2009, at least six were adult females, three of which were carrying near-term fetuses and four of which were just starting to bear calves (Waring *et al.* 2011). Since the average lifetime calf production is 5.25 calves (Fujiwara and Caswell 2001), the deaths of these six females represent a loss of reproductive potential of as many as 32 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 recorded calf over a 25-year period (Kraus *et al.* 2007). In contrast, one of the largest right whales on record, "Stumpy," as a prolific breeder, successfully rearing calves in 1980, 1987, 1990, 1993, and 1996 (Moore *et al.* 2007). Stumpy was killed in February 2004 of an apparent ship strike (NMFS 2006a). At the time of her death, she was estimated to be 30 years of age and carrying her sixth calf; the near-term fetus also died (NMFS 2006a).

Abundance estimates are an important part of assessing the status of the species. However, for section 7 purposes, the population trend (*i.e.*, whether increasing or declining) provides better information for assessing the effects of a proposed action on the species. As described in previous Opinions, data collected in the 1990s suggested that right whales were experiencing a slow but steady recovery (Knowlton *et al.* 1994). However, Caswell *et al.* (1999) used photo-identification data and modeling to estimate survival and concluded that right whale survival decreased from 1980 to 1994. Modified versions of the Caswell *et al.* (1999) model as well as several other models were reviewed at the 1999 IWC workshop (Best *et al.* 2001). Despite differences in approach, all of the models indicated a decline in right whale survival in the 1990s with female survival particularly affected (Best *et al.* 2001). In 2002, NMFS NEFSC hosted a workshop to review right whale population models to examine: (1) potential bias in the models, and (2) changes in the subpopulation trend based on new information collected in the late 1990s (Clapham *et al.* 2002). Three different models were used to explore right whale survivability and to address potential sources of bias. Although biases were identified that could negatively affect the results, all three modeling techniques resulted in the same conclusion: survival has continued to decline and seems to be affecting females disproportionately (Clapham *et al.* 2002). Increased mortalities in 2004 and 2005 were cause for serious concern (Kraus *et. al* 2005). Calculations indicate that this increased mortality rate would reduce population growth by approximately 10% per year (Kraus *et. al* 2005), in conflict with the 2.6% positive trend from 1990-2009 noted above by Waring *et al.* (2013). Despite the preceding, examination of the minimum number alive population index calculated from the individual sightings database for the years 1990-2009 suggest a positive and slowly accelerating trend in population size (Waring *et al.* 2013). These data reveal a significant increase in the number of catalogued right whales alive during this period (Waring *et al.* 2013). Recently, NMFS NEFSC developed a population viability analysis (PVA) to examine the influence of anthropogenic mortality reduction on the recovery prospects for the species (Pace, unpublished). The PVA evaluated how the populations would fare without entanglement mortalities as compared to the status quo. Only two of 1,000 projections (with the status quo simulation) ended with a smaller total population size than they started, and no projections resulted in extinction. As described above, the mean growth rate estimated in the latest stock assessment report was 2.6% (Waring *et al.* 2012). The potential biological removal (PBR)[13] for the Western Atlantic stock of North Atlantic right whale is 0.9 (Waring *et al.* 2013).

*Reproduction*
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 more than five years between 1998-2003, and then decreased to just over three 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 affecting right whales (Kraus *et al.* 2007), there is currently no evidence to support this. The dramatic reduction in the North Atlantic right whale population due to commercial whaling may have resulted in a loss of genetic diversity that could affect the ability of the current population to successfully reproduce (*i.e.*, decreased conceptions, increased abortions, and increased neonate

[13] Potential biological removal is the product of minimum population size, one-half the maximum net productivity rate and a "recovery" factor for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population.

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 underway to assess this relationship further and to examine 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. 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 PCB and DDT concentrations were lower than those found in other affected marine mammals (Weisbrod *et al.* 2000). Another suite of contaminants (i.e. antifouling agents and flame retardants) that disrupt reproductive patterns and have been found in other marine animals, raises new concerns (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008).

A number of diseases could be also affecting reproduction, although 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 diseases on right whales. Impacts of biotoxins on marine mammals are also poorly understood, yet there is some data 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 conclude that right whales are being exposed to measurable quantities of paralytic shellfish poisioning (PSP) toxins and domoic acid via trophic transfer from their prey upon which they feed (Durbin *et al.* 2002, Rolland *et al.* 2007).

Data on food-limitation are difficult to evaluate (Kraus *et al.* 2007). North Atlantic right whales seem to have thinner blubber than right whales from the South Atlantic (Kenney 2002; Miller *et al.* (2011). Miller *et al.* (2011) suggests that lipids in the blubber are used as energetic support for reproduction in female right whales. In the same study, blubber thickness was also compared among years of differing prey abundances. During a year of low prey abundance, right whales had significantly thinner blubber than during years of greater prey abundance. The results suggest that blubber thickness is indicative of right whale energy balance and that the marked fluctuations in the North Atlantic right whale reproduction have a nutritional component (Miller *et al.* (2011)).

Modeling work by Caswell *et al.* (1999) and Fujiwara and Caswell (2001) suggests that the North Atlantic Oscillation (NAO), a naturally occurring climatic event, affects the survival of mothers and the reproductive rate of mature females, and Clapham et al (2002) also suggests it affects calf survival. Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to 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 to 2001, consistent with the drops in copepod abundance. It has been hypothesized that right whale calving rates are a function of both food availability and 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*

Right whale recovery is negatively affected by anthropogenic mortality. From 2006 to 2010, right whales had the highest proportion relative to their population of reported entanglement and ship strike events of any species (Waring *et al.* 2012). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect on population growth rate than for other large whale species (Waring *et al.* 2012). For the period 2006-2010, the annual human-caused mortality and serious injury rate for the North Atlantic right whale averaged 3.0 per year (2.4 in U.S. waters; 0.6 in Canadian waters) (Waring *et al.* 2013). Nineteen confirmed right whale mortalities were reported along the U.S. East Coast and adjacent Canadian Maritimes from 2006 to 2010 (Henry *et al.* 2012). These numbers represent the minimum values for serious injury and 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 may generate false negatives 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.* 2012).

Considerable effort has been made to examine right whale carcasses for the cause of death (Moore *et al.* 2004). Examination is not always possible or conclusive because carcasses may be discovered floating at sea and cannot be retrieved, or may be in such an advanced stage of decomposition 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 be noted that mortality and serious injury event judgments are based upon the best available data and later information may result in revisions (Henry *et al.* 2012). Of the 19 total confirmed right whale mortalities (2006-2010) described in Henry *et al.* (2012), four were confirmed to be entanglement mortalities and five were confirmed to be ship strike mortalities. Serious injury involving right whales was documented for five entanglement events and one ship strike event.

Although disentanglement is often unsuccessful or not possible for many cases, there were at least two documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious injury from 2006 to 2010 (Waring *et al.* 2012). Even when entanglement or vessel collision does not cause direct mortality, it may weaken or compromise an individual so that subsequent injury or death is more likely (Waring *et. al* 2012). Some right whales that have been entangled were later involved in ship strikes (Hamilton *et al.* 1998) suggesting that the animal may have become debilitated by the entanglement to such an extent that it was less able to avoid a ship. Similarly, skeletal fractures and/or broken jaws sustained during a vessel collision may heal, but then compromise a whale's ability to efficiently filter feed (Moore *et al.* 2007). A necropsy of right whale #2143 ("Lucky") found dead in January 2005 suggested the animal (and her near-term fetus) died after healed propeller wounds from a 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.* 2012).

Entanglement records from 1990 to 2010 maintained by NMFS include 74 confirmed right whale entanglement events (Waring *et al.* 2012). 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.* 2012). 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 (142/202 females, 71.8%; 182/224 males, 81.3%), indicating that right whales of both sexes are equally vulnerable to entanglement. However, juveniles appear to become entangled at a higher rate than expected if all age groups were equally vulnerable. For all years but one (1998), the proportion of juvenile, entangled right whales exceeded their proportion within the population. Based on photographs of catalogued animals from 1935 through 1995, Hamilton *et al.* (1998) estimated that 6.4% of the North Atlantic right whale population exhibits signs of injury from vessel strikes.

Right whales are expected to be affected by climate change; however, no significant climate change-related impacts to right whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and the potential decline of forage.

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.

The indirect effects to right whales that may be associated with sea level rise are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to cetaceans is likely negligible.

The direct effects of increased $CO_2$ concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in marine plankton could have serious consequences for the marine food web.

*Summary of Right Whale Status*

In March 2008, NMFS listed the North Atlantic right whale as a separate, endangered species (*Eubalaena glacialis*) under the ESA. This decision was based on an analysis of the best scientific and commercial data available, taking 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) overuse 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, an October 2011 review of the photo-ID recapture database indicated that 444 individually recognized right whales were known to be alive in 2009 (Waring *et al.* 2013). The 2000/2001-2009/2010 calving seasons had relatively high calf production (31, 21, 19, 17, 28, 19, 23, 23, 39, and 19 calves, respectively) and included additional first time mothers (*e.g.*, eight new mothers in 2000/2001) (Waring *et al.* 2009, 2012).

Over the five-year period 2006-2010, 55 confirmed events involved right whales, 33 were confirmed entanglements and 13 were confirmed ship strikes. There were

19 verified right whale mortalities, four due to entanglements, and five due to ship strikes (Henry *et al.* 2012). This represents an absolute minimum number of the right whale mortalities for this period. Given the range and distribution of right whales in the North Atlantic, it is highly unlikely that all carcasses will be observed. Scarification analysis indicates that some whales do survive encounters with ships and fishing gear. However, the long-term consequences of these interactions are unknown. Right whale recovery is negatively affected by human causes of mortality. This mortality appears to, have a greater impact on the population growth rate of right whales, compared to other baleen whales in the western North Atlantic, given the small population size and low annual reproductive rate of right whales (Waring *et al.* 2012).

A variety of modeling exercises and analyses indicate that survival probability declined in the 1990s (Best *et al.* 2001), and mortalities in 2004-2005, including a number of adult females, also suggested an increase in the annual mortality rate (Kraus *et al.* 2005). Nonetheless, a census of the minimum number alive population index calculated from the individual sightings database as of October 21, 2011 for the years 1990-2009 suggest a positive trend in numbers of right whales (Waring *et al.* 2013). In addition, calving intervals appear to have declined to three years in recent years (Kraus *et al.* 2007), and calf production has been relatively high over the past several seasons.

#### 4.2.2 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 southern and northern 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 and are considered depleted under the MMPA. 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.* 2011). Although the IWC only considered one stock (Donovan 1991) there is evidence to indicate multiple populations migrating between their summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Angliss and Outlaw 2007, Carretta *et al.* 2011).

NMFS recognizes three management units within the U.S. EEZ in the Pacific for the purposes of managing this species under the MMPA. These are: the California-Oregon-Washington 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.* 2011). 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.* 2011). 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 that doubles previous population predictions (Calambokidis *et al.* 2008). There are indications that the California-Oregon-Washington stock was growing in the 1980s and early 1990s, with a best estimate of 8% growth per year (Carretta *et al.* 2011). The best available estimate for the California-Oregon-Washington stock is 2,043 whales (Carretta *et al.* 2011). The central North Pacific stock is estimated at 4,005 (Allen and Angliss 2011), and various studies report that it appears to have increased in abundance at rates between 6.6%-10% per year (Allen and Angliss 2011). Although there is no reliable population trend data for the western North Pacific stock, as surveys of the known feeding areas are incomplete and many feeding areas remain unknown, minimum population size is currently estimated at 732 whales (Allen and Angliss 2011).

The Northern Indian Ocean population of humpback whales consists of a resident stock in the Arabian Sea, which apparently does 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, residing 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 [60-111 95% confidence interval (CI)](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.* 2008). 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 parenthesis, 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.* 2008). 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.* 2008).

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

*Gulf of Maine (North Atlantic)*

Humpback whales from most Atlantic feeding areas calve and mate in the West Indies and migrate to feeding areas in the northwestern Atlantic during the summer months. Most of the humpbacks that forage in the Gulf of Maine visit Stellwagen Bank and the waters of Massachusetts and Cape Cod bays. Previously, the North Atlantic humpback whale population was treated as a single stock for management purposes, however due to the strong fidelity to the region displayed by many whales, the Gulf of Maine stock was reclassified as a separate feeding stock (Waring *et al.* 2012). 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 Jeffreys Ledge (CeTAP 1982) and peak in May and August. Small numbers of individuals may be present in this area, including the waters of Stellwagen Bank, year-round. They feed on small schooling fishes, particularly sand lance and Atlantic herring, targeting fish schools and filtering large amounts of water for their associated prey. Humpback whales may also feed on euphausiids (krill) as well as on capelin (Waring *et al.* 2010; Stevick *et al.* 2006).

In winter, whales from waters off New England, Canada, Greenland, Iceland, and Norway migrate to mate and calve primarily in the West Indies, where spatial and genetic mixing among these groups occurs (Waring *et al.* 2012). 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 1991a).

Humpback whales use the Mid-Atlantic as a migratory pathway to and from the calving/mating grounds, but it may also be an important winter feeding area for juveniles. Since 1989, observations of juvenile humpbacks in the Mid-Atlantic have been increasing during the winter months, peaking January through March (Swingle *et al.* 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean. Swingle *et al.* (1993) identified a shift in distribution of juvenile humpback whales in the nearshore waters of Virginia, primarily in winter months. Identified whales using the Mid-Atlantic area were found to be residents of the Gulf of Maine and Atlantic Canada (Gulf of St. Lawrence and Newfoundland) feeding groups, suggesting a mixing of different feeding populations in the Mid-Atlantic region. Strandings of humpback whales have increased between New Jersey and Florida since 1985, consistent with the increase in Mid-Atlantic whale sightings. Strandings between 1985 and 1992 were most frequent September through April in North Carolina and Virginia waters, and were composed primarily of juvenile humpback whales of no more than 11 meters in length (Wiley *et al.* 1995).

*Abundance Estimates and Trends*

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) (Stevick *et al.* 2003; Waring *et al.* 2013). 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.* 2012). The minimum population estimate for the Gulf of Maine stock is 823 whales, derived from a 2008 mark-recapture based count (Waring *et al.* 2013).

Population modeling, using data obtained from photographic mark-recapture studies, estimates the growth rate of the Gulf of Maine stock to be 6.5% for the period 1979-1991 (Barlow and Clapham 1997). More recent analysis for the period 1992-2000 estimated lower population growth rates ranging from 0% to 4.0%, depending on calf survival rate (Clapham *et al.* 2003 in Waring *et al.* 2012). However, it is unclear whether the apparent decline in growth rate is a bias result due to a shift in distribution documented for the period 1992-1995, or whether the population growth rates truly declined due to high mortality of young-of-the-year whales in U.S. Mid-Atlantic waters (Waring *et al.* 2012). Regardless, calf survival appears to have increased since 1996, presumably accompanied by an increase in population growth (Waring *et al.* 2012). Stevick *et al.* (2003) calculated an average population growth rate of 3.1% in the North Atlantic population overall for the period 1979-1993. The PBR for the Gulf of Maine stock of humpback whale is 2.7.

*Anthropogenic Injury and Mortality*

As 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 2006-2010, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 7.8 animals per year (U.S. waters, 7.2; Canadian waters, 0.6) (Waring *et al.* 2013). Between 2006 and 2010, humpback whales were involved in 101 confirmed entanglement events and 21 confirmed ship strike events (Henry *et al.* 2012). Over the five-year period, humpback whales were the most commonly reported entangled whale species; entanglements accounted for nine mortalities and 20 serious injuries (Henry *et al.* 2012). Of the 21 confirmed ship strikes, 10 of the events were fatal (Henry *et al.* 2012). It was assumed that all of these events involved members of the

Gulf of Maine stock of humpback whales unless a whale was confirmed to be from another stock. In reports prior to 2007, only events involving whales confirmed to be members of the Gulf of Maine stock were included. 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 (Henry *et al.* 2012; Waring *et al.* 2012).

Based on photographs taken from 2000-2002 of the caudal peduncle and fluke of humpback whales, Robbins and Mattila (2004) estimated that at least half (48-57%) of the sample (187 individuals) was coded as having a high likelihood of prior entanglement. Evidence suggests that entanglements have occurred at a minimum rate of 8-10% per year. Scars acquired by Gulf of Maine humpback whales between 2000 and 2002 suggest a minimum of 49 interactions with gear. Based on composite scar patterns, male humpback whales appear to be more vulnerable to entanglement than females. Males may be subject to other sources of injury that could affect scar pattern interpretation. Of the images obtained from a humpback whale breeding ground, 24% showed 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 Gulf of Maine male humpback whales (Robbins and Matilla 2004).

Humpback whales, like other baleen whales, may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources resulting from a variety of activities including fisheries operations, vessel traffic, and coastal development. Currently, there is no evidence that these types of activities are affecting humpback whales. However, Geraci *et al.* (1989) provide strong evidence that a mass mortality of humpback whales in 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. The occurrence of a red tide event may be related to an increase in freshwater runoff from coastal development, leading some observers to suggest that such events may become more common among marine mammals as coastal development continues (Clapham *et al.* 1999). There were three additional known cases of a mass mortality involving large whale species along the East Coast between 1998 and 2008. In the 2006 mass mortality event, 21 dead humpback whales were found between July 10 and December 31, 2006, triggering NMFS to declare an unusual mortality event (UME) for humpback whales in the Northeast United States. The UME was officially closed on December 31, 2007 after a review of 2007 humpback whale strandings and mortality showed that the elevated numbers were no longer being observed. The cause of the 2006 UME is listed as "undetermined," and the investigation has been closed, though could be re-opened if new information becomes available.

Changes in humpback whale 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.* 2012). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006). However, whether humpback whales were adversely affected by these trophic changes is unknown.

Humpback whales are expected to be affected by climate change; however, no significant climate change-related impacts to humpback whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and the potential decline of forage.

Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (MacLeod 2009). Humpback 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 indirect effects to humpback whales that may be associated with sea level rise are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). Cetaceans are unlikely to be directly affected by sea level rise, although important coastal bays for humpback breeding could be affected (IWC 1997).

The direct effects of increased $CO_2$ concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species.

*Summary of Humpback Whale Status*

The best available population estimate for humpback whales in the North Atlantic Ocean is 11,570 animals, and the best recent estimate for the Gulf of Maine stock is 823 whales (Waring *et al.* 2013). 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.* 2013). This is consistent with an estimated average trend of 3.1% in the North Atlantic population overall for the period 1979-1993 (Stevick *et al.* 2003). With respect to the species overall, there are also indications of increasing abundance for the California-Oregon-Washington, central North Pacific, and Southern Hemisphere stocks: Southwest Atlantic, Southeast Atlantic, Southwest Indian Ocean, Southeast Indian

Ocean, and Southwest Pacific. Trend data is lacking for the western North Pacific stock, the central South Pacific and Southeast Pacific subpopulations of the southern hemisphere humpback whales, and the northern Indian Ocean humpbacks.

#### 4.2.3 Fin Whale

The fin whale (*Balaenoptera physalus*) is listed as endangered under the ESA and also is designated as depleted under the MMPA. Fin whales inhabit a wide range of latitudes between 20-75°N and 20-75°S (Perry *et al.* 1999). The fin whale is ubiquitous in the North Atlantic and occurs from the Gulf of Mexico and Mediterranean Sea northward to the edges of the Arctic ice pack (NMFS 1998b). The overall pattern of fin whale movement is complex, consisting of a less obvious north-south pattern of migration than that of right and humpback whales. Based on acoustic recordings from hydrophone arrays, Clark (1995) reported a general southward flow pattern of fin whales in the fall from the Labrador/Newfoundland region, 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 gulping prey concentrations and filtering the water for the associated prey. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

*Pacific Ocean*
Within 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.* 2011). 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 the surveys covered only a portion of its range (Allen and Angliss 2010). An annual population increase of 4.8% between 1987-2003 was estimated for fin whales in coastal waters south of the Alaska Peninsula (Allen and Angliss 2010). This is the first estimate of population trend for North Pacific fin whales; however, it must be interpreted cautiously due to the uncertainty in the initial population estimate and the population structure (Allen and Angliss 2010). The best available estimate for the California/Washington/Oregon stock is 3,044, which is likely an underestimate (Carretta *et al.* 2011). The best available estimate for the Hawaii stock is 174, based on a 2002 line-transect survey (Carretta *et al.* 2011).

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

*North Atlantic*

NMFS has designated one population of fin whales in U.S. waters of the North Atlantic (Waring *et al.* 2012). This species is commonly found from Cape Hatteras northward. Researchers have suggested the existence of fin whale subpopulations in the North Atlantic based on local depletions resulting from commercial overharvesting (Mizroch and York 1984) or genetics data (Bérubé *et al.* 1998). Photo-identification studies in western North Atlantic feeding areas, particularly in Massachusetts Bay, have shown a high rate of annual return by fin whales, both within years and among years (Seipt *et al.* 1990) suggesting some level of site fidelity. The Scientific Committee of the International Whaling Commission (IWC) has proposed stock boundaries for North Atlantic fin whales. Fin whales off the eastern United States, Nova Scotia, and southeastern coast of Newfoundland are believed to constitute a single stock of fin whales under the present IWC scheme (Donovan 1991). However, it is uncertain whether the proposed boundaries define biologically isolated units (Waring *et al.* 2012).

During the 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.* 2012). Underwater listening systems have also demonstrated that the fin whale is the most acoustically common whale species heard in the North Atlantic (Clark 1995). The single most important area for this species appeared to be from the Great South Channel, along the 50 meter isobath past Cape Cod, over Stellwagen Bank, and past Cape Ann to Jeffreys 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 U.S. Mid-Atlantic coast from October through January suggest the possibility of an offshore calving area (Hain *et al.* 1992).

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

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

*Population Trends and Status*
Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort (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 U.S. continental shelf waters. The draft 2012 Stock Assessment Report (SAR) gives a best estimate of abundance for fin whales in the western North Atlantic of 3,522 (CV = 0.27). 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.* 2012). The minimum population estimate for the western North Atlantic fin whale is 2,817 (Waring *et al.* 2012). However, there are insufficient data at this time to determine population trends for the fin whale (Waring *et al.* 2012). The PBR for the western North Atlantic fin whale is 5.6.

Other estimates of the abundance of fin whales in the North Atlantic are presented in Pike *et al.* (2008) and Hammond *et al.* (2011). Pike *et al.* (2008) estimates the abundance of fin whales to be 27,493 (CV 0.2) in waters around Iceland and the Denmark Strait. Hammond *et al.* (2008) estimates the abundance of 19,354 (CV 0.24) fin whales in the eastern North Atlantic.

*Anthropogenic Injury and Mortality*
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 in U.S. and Canadian waters from 2006 to 2010 was 2.0 (U.S. waters, 1.8; Canadian waters, 0.2) (Waring *et al.* 2012). During this five-year period, there were 15 confirmed entanglements (two fatal; two serious injuries) and eight ship strikes (six fatal) (Henry *et al.* 2012). 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 an aboriginal subsistence whaling hunt for Greenland (Gambell 1993; Caulfield 1993). However, Iceland has increased its whaling activities in recent years and reported a catch of 136 whales in the 1988/89 and 1989/90 seasons (Perry *et al.* 1999), seven in 2006/07, and 273 in 2009/2010. Fin whales may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources resulting from a variety of activities.

Fin whales are expected to be affected by climate change; however, no significant climate change-related impacts to fin whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and the potential decline of forage.

Of the factors affecting geographic distribution of cetaceans, water temperature appears to be the main influence, with other factors primarily influencing how individuals are distributed within their ranges(MacLeod 2009). Cetacean species most likely to be affected by increases in water temperature are those with ranges restricted to non-tropical waters and with a preference for shelf waters. 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 indirect effects to fin whales that may be associated with sea level rise are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to fin whales is likely negligible.

The direct effects of increased $CO_2$ concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in marine plankton could have serious consequences for the marine food web.

*Summary of Fin Whale Status*

Information on the abundance and population structure of fin whales worldwide is limited. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the Southern Hemisphere is unknown and there are no current estimates of abundance for Southern Hemisphere fin whales. As noted above, the best population estimate for the western North Atlantic fin whale is 3,522 and the minimum population estimate is 2,817. The draft 2012 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, commercial whaling for fin whales in the North Atlantic has resumed and fin whales continue to be struck by large vessels. Based on the information currently available, for the purposes of this Opinion, NMFS considers the population trend for fin whales to be undetermined.

#### 4.2.4 Sei Whale

The sei whale (*Balaenoptera borealis*) is listed as endangered under the ESA and is designated as depleted under the MMPA. Sei whales are a widespread species in the world's temperate, subpolar, subtropical, and tropical marine waters. Sei whales reach sexual maturity at 5-15 years of age. The calving interval is believed to be two to three years (Perry *et al.* 1999).

*North Pacific and Southern Hemisphere*

The IWC only considers one stock of sei whales in the North Pacific (Donovan 1991), but for NMFS management purpose under the MMPA, sei whales within the Pacific U.S. EEZ are divided into three discrete non-contiguous areas: 1) waters around Hawaii, 2) California, Oregon, and Washington waters, and 3) Alaskan waters (Carretta *et al.* 2011). There are no abundance estimates for sei whales in the entire eastern North Pacific. The best estimate of abundance for California, Oregon, and Washington waters out to 300 nautical miles is 126 (CV=0.53) sei whales (Barlow and Forney 2007; Forney 2007; Carretta *et al.* 2011). No fishery related serious injuries or mortalities have been documented from 2004 through 2008 in the eastern North Pacific stock of sei whales (Carretta *et al.* 2011). During 2002-2008 there was one reported ship strike mortality in Washington in 2003 (NMFS Northwest Regional Office, unpublished data). The Hawaiian stock includes animals found both within the Hawaiian Islands EEZ and in adjacent international waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for international waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (Carretta *et al.* 2011). The best estimate of abundance for the Hawaiian stock of sei whales is 77 (CV=1.06). Between 2004 and 2008, no human-caused serious injury or mortality was documented in the Hawaiian stock of sei whales (Carretta *et al.* 2011).

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

*North Atlantic*

NMFS considers sei whales in the North Atlantic as one stock, known as the Nova Scotia stock (formerly known as the Western North Atlantic stock). Sei whales occur in deep water throughout their range, typically over the continental slope or in basins situated between banks (NMFS 1998b). In the Northwest Atlantic, it is speculated that the whales migrate from south of Cape Cod along the eastern Canadian coast in June and July, and return on a southward migration again in September and October (Waring *et al.* 2012). Olsen *et al.* (2009) tracked a tagged sei whale that moved from the Azores to off eastern Canada; however, such a migration remains unverified. 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. Recent springtime research in the Southwestern Gulf of Maine, suggests sei whales are reasonably common in this area in most years (Baumgartner *et al.* 2011).

Although sei whales may prey upon small schooling fish and squid, 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.* 2012). For purposes of the Marine Mammal Stock Assessment Reports, and based on a proposed IWC stock definition, NMFS recognizes the sei whales occurring from the U.S. East Coast to Cape Breton, Nova Scotia, and east to 42°W as the “Nova Scotia stock” of sei whales (Waring *et al.* 2012).

*Abundance Estimates and Trends*
The abundance estimate of 467 sei whales (CV=0.67), obtained from a shipboard and aerial survey conducted during June-August 2011, is considered the best available for the Nova Scotia stock of sei whales according to the draft 2012 SAR (Waring *et al.* 2012). This estimate is considered extremely conservative because all of the known range of this stock was not surveyed, and because of uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas. Hammond *et al.* (2011) estimates the abundance of sei whales in European Atlantic waters to be 619 (CV of 0.34) for identified sightings identified to species. The minimum population estimate for this sei whale stock is 279 (Waring *et al.* 2012). 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.* 2012). The PBR for the Nova Scotia stock sei whale is 0.6.

*Anthropogenic Injury and Mortality*
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 farther offshore than most commercial fishing operations, or perhaps entanglements do occur but are less likely to be observed. The minimum annual rate of confirmed human-caused serious injury and mortality to Nova Scotian sei whales from 2006 to 2010 was 1.2 (Waring *et al.* 2012), which includes 0.6 fishery interaction records and 0.6 vessel collision records. During this five-year period, there were three confirmed entanglements (one fatal; two serious injuries) and three ship strikes (all fatal) (Waring *et al.* 2012). Other impacts noted above for other baleen whales may also occur in this species (e.g., habitat degradation, etc.).

Sei whales are expected to be affected by climate change; however, no significant climate change-related impacts to sei whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats and the potential decline of forage.

Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (MacLeod 2009). Sei whales currently range from sub-polar to tropical waters. An increase in water temperature may be a favorable affect on sei whales, allowing them to expand their range into higher latitudes (MacLeod 2009).

The indirect effects to sei whales, that may be associated with sea level rise, are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to sei whales is likely negligible.

The direct effects of increased $CO_2$ concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in marine plankton could have serious consequences for the marine food web.

*Summary of Sei Whale Status*

The best estimate of abundance for the Nova Scotia stock of sei whales is 467 (Waring *et al.* 2012). There are insufficient data to determine trends of the Nova Scotian sei whale population. Two sei whale serious injuries and one mortality from fisheries interactions and three mortalities from ship strikes have been recorded in U.S. waters between 2006 and 2010 (Waring *et al.* 2012). Information on the status of sei whale populations worldwide is similarly lacking. There are no abundance estimates for sei whales in the entire eastern North Pacific, however the best estimate of abundance for California, Oregon, and Washington waters out to 300 nautical miles is 126 (Carretta *et al.* 2011). The stock structure of sei whales in the Southern Hemisphere is unknown. Based on the information currently available, for the purposes of this Opinion, NMFS considers the population trend for sei whales to be undetermined.

### 4.3 Status of Sea Turtles

Sea turtles continue to be affected by many activities occurring on the nesting beaches and in the marine environment. Poaching, habitat modification and destruction, and nesting predation affect eggs, hatchlings, and nesting females while on land. Fishery interactions, vessel interactions, marine pollution, and non-fishery operations (*e.g.*, dredging, military activities, oil and gas exploration), for example, affect sea turtles in the neritic zone, which is defined as the marine environment extending from mean low water down to 200 meters (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 meters (Lalli and Parsons 1997).[14] 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.

Leatherback, Kemp's ridley, and green sea turtles are listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS), while loggerhead sea turtles are listed by DPS. Information on the range-wide status of each species is included, where appropriate. Additional background information on the range-wide status of these species, as well as a description and life history of the species, can be found in a number of published documents, including sea turtle status reviews and biological reports (NMFS and USFWS 1995; Hirth 1997; Turtle Expert Working Group [TEWG] 1998, 2000, 2007, 2009; NMFS and USFWS 2007a, 2007b, 2007c, 2007d), and recovery plans for the loggerhead sea turtle (NMFS and USFWS 1998a, 2008), leatherback sea turtle (NMFS and USFWS 1992b, 1998b), Kemp's ridley sea turtle (NMFS and USFWS 1992a), and green sea turtle (NMFS and USFWS 1991, 1998c).

### 4.3.1 NWA DPS of Loggerhead Sea Turtle

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

Listing History

Loggerhead sea turtles were listed as threatened throughout their global range on July 28, 1978. Since that time, several status reviews have been conducted to review the status and recommendations have been made regarding its ESA listing status. Based on a 2007 five-year status review of the species, which discussed the range of threats to loggerheads including climate change, NMFS and USFWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, the 2007 status review also determined that an analysis and review of the species should be conducted to determine whether DPSs should be identified for the loggerhead sea turtle (NMFS and USFWS 2007a). This initiative was supported by studies showing that genetic differences exist between loggerhead

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

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

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

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

On March 16, 2010, NMFS and USFWS published a proposed rule (75 FR 12598) that would divide the worldwide population of loggerhead sea turtles into nine DPSs, as described in the 2009 Status Review. Two of the DPSs were proposed to be listed as threatened and seven of the DPSs, including the Northwest Atlantic Ocean DPS, were proposed to be listed as endangered. NMFS and USFWS accepted comments on the proposed rule through September 13, 2010 (75 FR 30769, June 2, 2010). On March 22, 2011 (76 FR 15932), NMFS and USFWS extended the date by which a final determination would be made and solicited new information and analysis. This action was taken to address the interpretation of the existing data on status and trends and its relevance to the assessment of risk of extinction for the Northwest Atlantic Ocean DPS, as well as the magnitude and immediacy of the fisheries bycatch threat and measures to reduce this threat.

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

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

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

*al.* 2006; Revelles *et al.* 2007). Previous literature (Bowen *et al.* 2004) has suggested that there is the potential, albeit small, for some juveniles from the Mediterranean DPS to be present in U.S. Atlantic coastal foraging grounds. These conclusions must be interpreted with caution, however, as they may be representing a shared common haplotype and lack of representative sampling at Eastern Atlantic rookeries rather than an actual presence of Mediterranean DPS turtles in U.S. Atlantic coastal waters. A re-analysis of the data by the Atlantic loggerhead Turtle Expert Working Group has found that that it is unlikely that U.S. fishing fleets are interacting with either the Northeast Atlantic loggerhead DPS or the Mediterranean loggerhead DPS (LaCasella *et al.* In Review). Given that the action area is a subset of the area fished by U.S. fleets, it is reasonable to assume that, based on this new analysis, no individuals from the Mediterranean DPS or Northeast Atlantic DPS would be present in the action area. Sea turtles of the South Atlantic DPS do not inhabit the action area of this consultation (Conant *et al.* 2009). The remainder of this consultation will only focus on the NWA DPS, listed as threatened.

<u>Distribution and Life History</u>
Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitats and foraging areas for loggerheads within the Atlantic Ocean. Detailed information is also provided in the five-year status review for loggerheads (NMFS and USFWS 2007a), the TEWG report (2009), and the final revised Recovery Plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2008).

In the western Atlantic, waters as far north as 41°N to 42°N 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, MA and in the Gulf of Mexico from Florida to Texas, although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a, 1995b; Braun and Epperly 1996; Braun-McNeill *et al.* 2008; Mitchell *et al.* 2003). Loggerheads have been observed in waters with surface temperatures of 7°C to 30°C, but water temperatures ≥11°C are most favorable (Shoop and Kenney 1992; Epperly *et al.* 1995b). The presence of loggerhead sea turtles in U.S. Atlantic waters is also influenced by water depth. Surveys of continental shelf waters north of Cape Hatteras, NC indicated that loggerhead sea turtles were most commonly sighted in waters with bottom depths ranging from 22 to 49 meters deep (Shoop and Kenney 1992). Loggerheads were observed in waters ranging in depth from 0 (i.e., on the beach) to 14,701 feet (4,481 meters) (Shoop and Kenney 1992). More recent survey and satellite tracking data support that they occur in waters from the beach to beyond the continental shelf (Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; Mansfield 2006; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield *et al.* 2009).

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

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

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

Table 11 (taken from the 2008 loggerhead recovery plan) highlights the key life history parameters for loggerheads nesting in the U.S.

**Table 11** Typical values of life history parameters for loggerheads nesting in the U.S.

| Life History Parameter | Data |
|---|---|
| Clutch Size | 100-126 eggs[15] |

[15] Dodd (1988).

| | |
|---|---|
| Egg incubation duration (varies depending on time of year and latitude) | 42-75 days[16,17] |
| Pivotal temperature (incubation temperature that produces an equal number of males and females) | 29.0˚C[18] |
| Nest productivity (emerged hatchlings/total eggs) x 100 (varies depending on site specific factors) | 45-70%[2,3] |
| Clutch frequency (number of nests/female/season) | 3-5.5 nests[19] |
| Internesting interval (number of days between successive nests within a season) | 12-15 days[20] |
| Juvenile (<87 cm CCL) sex ratio | 65-70%[21] |
| Remigration interval (number of years between successive nesting migrations) | 2.5-3.7 years[22] |
| Nesting season | Late April-Early September |
| Hatching season | Late June-early November |
| Age at sexual maturity | 32-25 years[23] |
| Life span | >57 years[24] |

Population Dynamics and Status

The majority of Atlantic nesting occurs on beaches of the southeastern United States (NMFS and USFWS 2007a). For the past decade, 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 ; (2) a south Florida group of nesting females that nest from 29°N 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

[16] Dodd and Mackinnon (1999, 2000, 2001, 2002, 2003, 2004).
[17] Blair Witherington, FFWCC, personal communication, 2006 (information based on nests monitored throughout Florida beaches in 2005, n=865).
[18] Mrosovsky (1988).
[19] Murphy and Hopkins (1984); Frazer and Richardson (1985); Ehrhart, unpublished data; Hawkes *et al.* (2005); Scott (2006); Tony Tucker, Mote Marine Laboratory, personal communication (2008).
[20] Caldwell (1962); Dodd (1988).
[21] National Marine Fisheries Service (2001); Allen Foley, FFWCC, personal communication (2005).
[22] Richardson *et al.* (1978); Bjorndal *et al.* (1983); Ehrhart, unpublished data.
[23] Melissa Snover, NMFS, personal communication (2005).
[24] Dahlen *et al.* (2000).

Peninsula, Mexico; and (5) a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key West, FL and on Cal Sal Bank (TEWG 2009). Genetic analyses of mitochondrial DNA, which a sea turtle inherits from its mother, indicate that there are genetic differences between loggerheads that nest at and originate from the beaches used by each of the five identified nesting groups of females (TEWG 2009). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both parents, indicates little to no genetic differences between loggerheads originating from nesting beaches of the five Northwest Atlantic nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site fidelity to nesting beaches within a particular area, while males provide an avenue of gene flow between nesting groups by mating with females that originate from different nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow, however, is unclear (Shamblin 2007).

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

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

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

NMFS and USFWS (2008), Witherington *et al.* (2009), and TEWG (2009) analyzed the status of the nesting assemblages within the NWA DPS using standardized data collected over periods ranging from 10 to 23 years. These analyses used different analytical approaches, but all found that there had been a significant overall nesting decline within the NWA DPS. However, with the addition of nesting data from 2008 to 2012, the trend line changes, showing a strong positive trend since 2007 (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). The nesting data presented in the Recovery Plan (through 2008) are described below, with updated trend information through 2010 for two recovery units.

From the beginning of standardized index surveys in 1989 until 1998, the PFRU, the largest nesting assemblage in the Northwest Atlantic by an order of magnitude, had a significant increase in the number of nests. However, from 1998 through 2008, there was a 41% decrease in annual nest counts from index beaches, which represent an average of 70% of the statewide nesting activity (NMFS and USFWS 2008). From 1989 to 2008, the PFRU had an overall declining nesting trend of 26% (95% CI: -42% to -5%; NMFS and USFWS 2008). With the addition of nesting data through 2010, the nesting trend for the PFRU does not show a nesting decline statistically different from zero (76 FR 58868, September 22, 2011).

The NRU, the second largest nesting assemblage of loggerheads in the United States, has been declining at a rate of 1.3% annually since 1983 (NMFS and USFWS 2008). The trend was analyzed using nesting data available as of October 2008. The NRU dataset included 11 beaches with an uninterrupted 20-year time series; these beaches represent approximately 27% of NRU nesting in 2008. Through 2008, there was strong statistical data to suggest the NRU has experienced a long-term decline, but with the inclusion of nesting data through 2010, nesting for the NRU is showing possible signs of stabilizing (76 FR 58868, September 22, 2011).

Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. However, the NGMRU has shown a significant declining trend of 4.7% annually since index nesting beach surveys were initiated in 1997 (NMFS and USFWS 2008). The trend was analyzed using nesting data available as of October 2008.

No statistical trends in nesting abundance can be determined for the DTRU because of the lack of long-term data. Similarly, statistically valid analyses of long-term nesting trends for the entire GCRU are not available because there are few long-term standardized nesting surveys representative of the region. Additionally, changing survey effort at monitored beaches and scattered and low-level nesting by loggerheads at many locations currently precludes comprehensive analyses (NMFS and USFWS 2008).

Sea turtle census nesting surveys are important in that they provide information on the relative abundance of nesting each year, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 2008 Recovery Plan compiled information on mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (*i.e*., nesting groups). They are: (1) for the NRU, a mean of 5,215 loggerhead nests per year (1989-2008) with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year (1989-2007) with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year (1995-2004, excluding 2002) with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year (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 (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 estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. The above values for average nesting females per year were based upon 4.1 nests per female per Murphy and Hopkins (1984).

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

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

Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, SC to St. Augustine, FL) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the 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 four years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977 to 2004 show an increasing trend of loggerheads at the 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 compared to the period 1987-1992. Only two loggerheads (of a total 54 turtles) were observed captured in pound net gear during the period 2002-2004, while the previous decade's study recorded 11 to 28 loggerheads per year (Morreale *et al.* 2005). No additional loggerheads were reported captured in pound net gear in New York through 2007, although two were found cold-stunned on Long Island Bay beaches in the fall of 2007 (Memo to the File, L. Lankshear, December 2007). Potential explanations for this decline include major shifts in loggerhead foraging areas and/or increased mortality in pelagic or early benthic stage/age classes (Morreale *et al.* 2005). Using aerial surveys, Mansfield (2006) also found a decline in the densities of loggerhead sea turtles in Chesapeake Bay over the period 2001-2004 compared to aerial survey data collected in the 1980s. Significantly fewer loggerheads ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to those observed during aerial surveys in the 1980s (Mansfield 2006). A comparison of median densities from the 1980s to the 2000s suggested that there had been a 63.2% reduction in densities during the spring residency period and a 74.9% reduction in densities during the summer residency period (Mansfield 2006). The decline in observed loggerhead populations in Chesapeake Bay may be related to a significant decline in prey, namely horseshoe crabs and blue crabs, with loggerheads redistributing outside of Bay waters (NMFS and USFWS 2008).

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

As part of the Atlantic Marine Assessment Program for Protected Species (AMAPPS), line transect aerial abundance surveys and turtle telemetry studies were conducted along the Atlantic Coast and annual reports for 2010 and 2011 have been produced. AMAPPS is a multi-agency initiative to assess marine mammal, sea turtle, and seabird abundance and distribution in the Atlantic. As presented in NMFS NEFSC (2011a), the 2010 survey found a preliminary total surface abundance estimate within the entire study area of about 60,000 loggerheads (CV=0.13) or 85,000, if a portion of unidentified hard-shelled sea turtles were included (CV=0.10). Surfacing times were generated from the satellite tag data collected during the aerial survey period, resulting in a 7% (5%-11% inter-quartile range) median surface time in the South Atlantic area and a 67% (57%-77% inter-quartile range) median surface time to the north. The calculated preliminary regional abundance estimate is about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NMFS NEFSC 2011a). The estimate increases to approximately 801,000 (inter-quartile range of 521,000-1,111,000) when based on known loggerheads and a portion of unidentified turtle sightings. The density of loggerheads was generally lower in the north than the south; based on number of turtle groups detected, 64% were seen south of Cape Hatteras, NC, 30% in the southern Mid-Atlantic Bight, and 6% in the northern Mid-Atlantic Bight. Although they have been seen farther north in previous studies (*e.g.*, Shoop and Kenney 1992) and in the 2011 AMAPPS surveys, no loggerheads were observed during the aerial surveys conducted in the summer of 2010 in the more northern zone encompassing Georges Bank, Cape Cod Bay, and the Gulf of Maine (NMFS NEFSC 2011a). These estimates of loggerhead abundance over the U.S. Atlantic continental shelf are considered very preliminary. A more thorough analysis will be completed pending the results of further studies related to improving estimates of regional and seasonal variation in loggerhead surface time (by increasing the sample size and geographical area of tagging) and other information needed to improve the biases inherent in aerial surveys of sea turtles (*e.g.*, research on depth of detection and species misidentification rate). This survey effort represents the most comprehensive assessment of sea turtle abundance and distribution in many years. Additional results from aerial surveys and research to improve the abundance estimates are anticipated for 2012-2014, depending on available funds.

Threats

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

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

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

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

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

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

In addition to improvements in TED design, interactions between loggerheads and the shrimp fishery had been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 South Atlantic and GOM Shrimp Opinion (NMFS 2002a) take estimates were based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of 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 were substantially less than were projected in the 2002 Opinion. In 2008, the NMFS Southeast Fisheries Science Center (SEFSC) estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery to be 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B.

Ponwith, Southeast Fisheries Science Center to Dr. R. Crabtree, Southeast Region, PRD, December 2008). In August 2010, NMFS reinitiated section 7 consultation on southeastern state and federal shrimp fisheries based on a high level of strandings, elevated nearshore sea turtle abundance as measured by trawl catch per unit of effort, and lack of compliance with TED requirements. The 2012 section 7 consultation on the shrimp fishery was unable to estimate the current total annual level of take for loggerheads. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least thousands and possibly tens of thousands of interactions annually, of which at least hundreds and possibly thousands are expected to be lethal (NMFS 2012a).

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

There have been several published estimates of the number of loggerheads taken annually as a result of the dredge fishery for Atlantic sea scallops, ranging from a low of zero in 2005 (Murray 2007) to a high of 749 in 2003 (Murray 2004). Murray (2011) recently evaluated loggerhead sea turtle interactions in scallop dredge gear from 2001 to 2008. In that paper, the average number of annual observable interactions of hard-shelled sea turtles in the Mid-Atlantic scallop dredge fishery prior to the implementation of chain mats (January 1, 2001 through September 25, 2006) was estimated to be 288 turtles (CV = 0.14, 95% CI: 209-363) [equivalent to 49 adults], 218 of which were loggerheads [equivalent to 37 adults]. After the implementation of chain mats, the average annual number of observable interactions was estimated to be 20 hard-shelled sea turtles (CV = 0.48, 95% CI: 3-42), 19 of which were loggerheads. If the rate of observable interactions from dredges without chain mats is applied to trips with chain mats, the estimated number of observable and inferred interactions of hard-shelled sea turtles after chain mats were implemented is 125 turtles per year (CV = 0.15, 95% CI: 88-163) [equivalent to 22 adults], 95 of which were loggerheads [equivalent to 16 adults]. Interaction rates of hard-shelled turtles were correlated with sea surface temperature, depth, and use of a chain mat. Results from this recent analysis suggest that chain mats and fishing effort reductions have contributed to the decline in estimated loggerhead sea turtle interactions with scallop dredge gear after 2006 (Murray 2011). Turtle Deflector Dredges (TDDs) are required in the scallop fishery as of May 1, 2013, and are expected to further decrease serious injuries to sea turtles.

An estimate of the number of loggerheads taken annually in U.S. Mid-Atlantic gillnet fisheries has also recently been published (Murray 2009a, b). From 1995 to 2006, the annual bycatch of loggerheads in U.S. Mid-Atlantic gillnet gear was estimated to average 350 turtles (CV=0.20, 95% CI over the 12-year period: 234 to 504). Bycatch rates were correlated with latitude, sea surface temperature, and mesh size. The highest predicted bycatch rates occurred in warm waters of the southern Mid-Atlantic in large-mesh (>7 inch/17.8 cm) gillnets (Murray 2009a). In the spring of 2000, a total of 275 loggerhead carcasses were recovered from North Carolina beaches. The cause of death for most of the turtles was unknown, but NMFS suspects that the mass mortality event was caused by a large-mesh gillnet fishery for monkfish and dogfish operating offshore in the preceding weeks (67 FR 71895, December 3, 2002).

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species (HMS) FMP are estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). NMFS has mandated gear changes for the HMS fishery to reduce sea turtle bycatch and the likelihood of death from those incidental takes that would still occur (Garrison and Stokes 2012). In 2010, there were 40 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2012). All of the loggerheads were released alive, with 29 out of 40 (72.5%) released with all gear removed. A total of 344.4 (95% CI: 236.6-501.3) loggerhead sea turtles were estimated to have interacted with the longline fisheries managed under the HMS FMP in 2010 based on the observed bycatch events (Garrison and Stokes 2012). The 2010 estimate is considerably lower than those in 2006 and 2007 and is well below the historical highs that occurred in the mid-1990s (Garrison and Stokes 2012). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Documented interactions also occur in other fishery gear types and by non-fishery mortality sources (*e.g.*, hopper dredges, power plants, vessel collisions), although quantitative/qualitative estimates are only available for activities on which NMFS has consulted (See sections 5.1.3, 5.1.4 and 5.1.5 below).

The most recent Recovery Plan for loggerhead sea turtles as well as the 2009 Status Review Report identifies global climate change as a threat to loggerhead sea turtles. For a complete discussion of how globale climate change may affect the NWA loggerhead DPS, see Section 6.0.

*Summary of Status for Loggerhead Sea Turtles*
Loggerheads are a long-lived species and reach sexual maturity at around 32-35 years in the Northwest Atlantic (NMFS and USFWS 2008). The species continues to be affected by many factors occurring on nesting beaches and in the water. These include poaching, habitat loss, and nesting predation that affects eggs, hatchlings, and nesting females on land, as well as fishery interactions, vessel interactions, marine pollution, and non-fishery (*e.g.*, dredging) operations affecting all sexes and age classes in the water (NRC 1990; NMFS and USFWS 2007a, 2008). As a result, loggerheads still face many of the original threats that were the cause of their listing under the ESA. Of the nine DPSs defined in the NMFS and USFWS final rule (75 FR 12598), only the NWA DPS is considered in this Opinion.

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

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

While several documents reported the decline in nesting numbers in the NWA DPS (NMFS and USFWS 2008, TEWG 2009), when nest counts through 2012 are analyzed, researchers found no demonstrable trend, indicating a reversal of the post-1998 decline (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). The SEFSC (2009) estimated the number of adult females in the NWA DPS at 30,000, and if a 1:1 adult sex ratio is assumed, the result is 60,000 adults in this DPS. Based on the reviews of nesting data, as well as information on population abundance and trends, NMFS and USFWS determined in the September 2011 listing rule that the NWA DPS should be listed as threatened. They found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats.

Based on the information presented above, for purposes of this Opinion, we consider that the status of NWA DPS of loggerheads over the next ten years will be no worse than it is currently. Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of pelagic immatures, benthic immatures, and sexually mature age classes from various fisheries and other marine activities (Conant *et al.* 2009). Recent actions have taken significant steps towards reducing the recurring sources of mortality and improving the status of all nesting stocks. For example, TED, chain mat, and TDD regulations represent a significant improvement in the baseline effects of trawl and dredge fisheries on loggerheads in the Northwest Atlantic, although shrimp trawling is still considered to be one of the largest sources of anthropogenic mortality on loggerheads (SEFSC 2009, NMFS 2012h). Loggerhead nesting has been on the rise since 2008, and Van Houton and Halley (2011) suggest that nesting in Florida, which contains by far the largest loggerhead rookery in the DPS, could substantially increase over the next few decades.

### 4.3.2 Leatherback Sea Turtle

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

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

*Pacific Ocean*

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

The largest extant leatherback nesting group in the Indo-Pacific lies on the North Vogelkop coast of West Papua, Indonesia, with 3,000-5,000 nests reported annually in the 1990s (Suárez *et al.* 2000). However, in 1999, local villagers started reporting dramatic declines in sea turtles near their villages (Suárez 1999). Declines in nesting groups have been noted throughout the western Pacific region, where observers report that nesting groups are well below abundance levels observed several decades ago (*e.g.*, Suárez 1999). Leatherback sea turtles in the western Pacific are threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals.

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

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

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

*Indian Ocean*

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

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

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

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

The CeTAP aerial survey of the outer Continental Shelf from Cape Hatteras, NC 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 meters, but 84.4% of sightings were in waters less than 180 meters (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 than loggerhead sea turtles since more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). Studies of satellite-tagged leatherbacks suggest that they spend 10-41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005b). The greatest amount of surface time (up to 41%) was recorded when leatherbacks occurred in continental shelf and slope waters north of 38°N (James *et al.* 2005b).

In 1979, the waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands were designated as critical habitat for the leatherback sea turtle. NMFS is currently reviewing whether the addition of waters adjacent to a major nesting beach in Puerto Rico to the critical habitat designation is warranted. USFWS also plans to address this region during a future planned status review. On February 2, 2010, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters adjacent to a major nesting beach in Puerto Rico. NMFS published a 90-day finding on July 16, 2010, which found that the petition did not present substantial scientific information indicating that the revision was warranted. The original petitioners submitted a second petition on November 2, 2010 to revise the critical habitat designation to include waters adjacent to a major nesting beach in Puerto Rico, and this time included additional information on the usage of the waters. On May 5, 2011, NMFS determined that a revision to critical habitat off Puerto Rico may be warranted, but on June 4, 2012 issued a decision denying the petition due to a lack of reasonably defined physical or biological features that are essential to the leatherback sea turtle's conservation and that may require special management considerations or protection (77 FR 32909). Note that on August 4, 2011, USFWS issued a determination that revision to critical habitat along Puerto Rico should be made and will be addressed during the future planned status review.

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

<u>Population Dynamics and Status</u>

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

In the U.S., the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b) to 1,712 recorded in 2012 (FWC 2013). Stewart *et al.* (2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. An analysis of Florida's index nesting beach sites from 1989 to 2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (TEWG 2007). The TEWG reports an increasing or stable nesting trend for five of the seven populations or groups of populations, with the exceptions of the Western Caribbean and West Africa groups. The leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (TEWG 2007), and represents more than half of total nesting by leatherback sea turtles worldwide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group also seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests in Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The TEWG (2007) report indicates that a positive population growth rate was found for French Guinea and Suriname using nest numbers from 1967 to 2005, a 39-year period, and that there was a 95% probability that the population was growing. Given the magnitude of leatherback nesting in this area compared to other nest sites, negative impacts in leatherback sea turtles in this area could have profound impacts on the entire species.

The CeTAP aerial survey conducted from 1978 to 1982 estimated the summer leatherback population for the northeastern United States at approximately 300-600 animals (from near Nova Scotia, Canada to Cape Hatteras, North Carolina) (Shoop and Kenney 1992). However, the estimate was based on turtles visible at the surface and does not include those that were below the surface out of view. Therefore, it likely underestimated the leatherback population. 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 suggested that the true abundance of leatherbacks may be 4.27 times higher (Palka 2000).

<u>Threats</u>

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

Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer-reviewed publications and NMFS documents (*e.g.*, biological opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually after implementation of bycatch mitigation measures. Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for leatherbacks at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in a few hundred interactions annually, of which a subset are expected to be lethal (NMFS 2012a).

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

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990 to 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). From 2002 to 2011, NMFS received 159 reports of sea turtles entangled in vertical lines from Maine to Virginia, with 147 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 147 confirmed events during this period, 133 events involved leatherbacks, 13 involved loggerheads, and 1 invovled a green sea turtle. NMFS identified the gear type and fishery for 93 of the 147 confirmed events, which included lobster (51[25]), whelk/conch (23), black sea bass (10), crab (7), and research pot gear (2). A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.* 2002).

Leatherback interactions with the U.S. South Atlantic and Gulf of Mexico shrimp fisheries are also known to occur (NMFS 2002a). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, FL through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003, to amend the TED regulations (68 FR 8456, February 21, 2003). Modified TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles. With these gear modifications, Epperly *et al.* (2002) anticipated an average of 80 leatherback mortalities a year in shrimp gear interactions, but dropped

[25] One case involved both lobster and whelk/conch gear, but this animal is listed only under the lobster group.

the estimate to 26 leatherback mortalities in 2009 due to effort reduction in the southeast shrimp fishery (Memo from Dr. B. Ponwith, SEFSC, to Dr. R. Crabtree, SERO, January 5, 2011). The most recent Opinion, issued in 2012, does not give a numerical ITS for leatherbacks, but instead monitors TED compliance and fishery effort to monitor and limit take (NMFS 2012).

Other trawl fisheries are also known to interact with leatherback sea turtles on a much smaller scale. For example, NMFS fisheries observers documented leatherbacks taken in trips targeting *Loligo* squid off Delaware in 2001 and off Connecticut in 2009, and targeting little skate off Connecticut in 2011. 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. NEFOP data from 1994 to 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in pelagic drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 64% to 99% (Waring *et al.* 2000). 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 Cape Hatteras (STSSN unpublished data reported in NMFS SEFSC 2001). Murray (2009a) reports five observed leatherback captures in Mid-Atlantic sink gillnet fisheries between 1994 and 2008.

Fishing gear interactions can occur throughout the leatherback's range, including in Canadian waters. Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in salmon nets, herring nets, gillnets, trawl lines, and crab pot lines. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline seen in the leatherback sea turtle population in French Guiana from 1973 to 1998 (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill sea turtles in the waters of coastal Nicaragua also incidentally catch leatherback sea turtles (Lagueux *et al.*1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off Trinidad and Tobago, with mortality estimated to be between 50% and 95% (Eckert and Lien 1999). Many of the sea turtles do not die as a result of drowning, but rather because the fishermen butcher them to remove them from their nets (NMFS SEFSC 2001).

Leatherbacks may be more susceptible to marine debris ingestion than other sea turtle species due to the tendency of floating debris to concentrate in convergence zones that juveniles and adults use for feeding (Shoop and Kenney 1992; Lutcavage *et al.* 1997). Investigations of the necropsy results of leatherback sea turtles revealed that a substantial percentage (34% of the 408 leatherback necropsies recorded between 1985 and 2007) reported plastic within the turtles' stomach contents, and in some cases (8.7% of cases in which plastic was reported), blockage of the gut may have caused the mortality (Mrosovsky *et al.* 2009). An increase in reports of plastic ingestion was evident in leatherback necropsies conducted after the late 1960s (Mrosovsky *et al.* 2009). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items (*e.g.*, jellyfish) and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that plastic objects may resemble food items by their shape, color, size, or drifting movements, and induce a feeding response in leatherbacks.

Global climate change has been identified as a factor that may affect leatherback habitat and biology (NMFS and USFWS 2007b); however, no significant climate change related impacts to leatherback sea turtle populations have been observed to date. Over the long term, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Changes in marine systems associated with rising water temperatures, changes in ice cover, salinity, oxygen levels and circulation including shifts in ranges and changes in algal, plankton, and fish abundance could affect leatherback prey distribution and abundance. Climate change is expected to expand foraging habitats into higher latitude waters and some concern has been noted that increasing temperatures may increase the female:male sex ratio of hatchlings on some beaches (Mrosovsky *et al.* 1984 and Hawkes *et al.* 2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b). Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al.* 2009). Leatherbacks have expanded their range in the Atlantic north by 330 kilometers in the last 17 years as warming has caused the northerly migration of the 15°C SST isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary jellyfish prey, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al.* 1999; Attrill *et al.* 2007; Richardson *et al.* 2009). However, any increase in jellyfish populations may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited.

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

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

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

#### 4.3.3 Kemp's Ridley Sea Turtle

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

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

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

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

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

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

*Population Dynamics and Status*

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

*Threats*

Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, predators, and oceanographic-related events such as cold-stunning. Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that use the more northern habitats of Cape Cod Bay and Long Island Sound. In the last six years (2007-2013), the number of cold-stunned turtles ranged from a low in 2007 of 66 (40 Kemp's ridleys, seven loggerheads, 16 greens, and three unknown) to a high in 2013 of 491 (273 Kemp's ridleys, 167 loggerheads, 43 greens, and eight unknown). Annual cold stunning events vary in magnitude; the magnitude of episodic major cold stunning events may be associated with numbers of turtles using northeast U.S.

waters in a given year, oceanographic conditions, and/or the occurrence of storm events in the late fall. Although many cold-stunned turtles can survive if they are found early enough, these events are a significant source of natural mortality for Kemp's ridleys.

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

Although modifications to shrimp trawls have helped to reduce mortality of Kemp's ridleys, a recent assessment found that the Southeast/Gulf of Mexico shrimp trawl fishery remained responsible for the vast majority of U.S. fishery interactions (up to 98%) and mortalities (more than 80%). Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer-reviewed publications and NMFS documents (*e.g.,* biological opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, has occurred annually after implementation of bycatch mitigation measures. Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The 2012 section 7 consultation on the shrimp fishery was unable to estimate the total annual level of take for Kemp's ridleys at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least tens of thousands and possibly hundreds of thousands of interactions with Kemp's ridleys annually, of which at least thousands and possibly tens of thousands are expected to be lethal (NMFS 2012a).

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

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

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

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

As with the other sea turtle species discussed in this section, while there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al.* 2009). Based on the most recent five-year status review (NMFS and USFWS 2007c), and following from the climate change discussions on loggerheads and leatherbacks, it is unlikely that impacts from climate change will have a significant effect on the status of Kemp's ridleys over the scope of the action assessed in this Opinion, which is the next ten years. However, significant impacts from climate change in the future beyond ten years are to be expected, but the severity of and rate at which these impacts will occur is currently unknown.

*Summary of Status for Kemp's Ridley Sea Turtles*

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

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

#### 4.3.4 Green Sea Turtles

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

*Pacific Ocean*

Green sea turtles occur in the western, central, and eastern Pacific. Foraging areas are located throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998c). In the western Pacific, major nesting rookeries at four sites including Heron Island (Australia), Raine Island (Australia), Guam, and Japan were evaluated. Three where determined to be increasing in abundance, while the population in Guam appears stable (NMFS and USFWS 2007d). In the central Pacific, nesting occurs on French Frigate Shoals, HI, which has also been reported as increasing, with a mean of 400 nesting females annually from 2002 to 2006 (NMFS and USFWS 2007d). In 2012, we received a petition to delist the Hawaiian population of green sea turtles, and our 90-day finding determined that the petition, viewed in context of information readily available in our files, presents substantial scientific and commercial information indicating that the petition action may be warranted (77 FR 45571). A status review is currently underway. The main nesting sites for green sea turtles in the eastern Pacific are located in Michoacan, Mexico and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007d). However, historically, more than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). The Pacific Mexico green turtle nesting population (also called the black turtle) is considered endangered.

Historically, green sea turtles were caught for food in many areas of the Pacific. They also were commercially exploited, which, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 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). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green sea turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the recent past, only the Comoros Island Index Site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

*Mediterranean Sea*

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

*Atlantic Ocean*

<u>Distribution and Life History</u>

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

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

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

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

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

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

By far, the most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970s and nest count data from 1999 to 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 Trindad Island, Brazil number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007d).

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

The statewide Florida surveys (2000-2012) have show an increasing trend of green sea turtle nesting, with a low of 581 in 2001 to a high of 15,352 in 2011 (NMFS and USFWS 2007d, FWC 2013). 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, and Florida panhandle beaches (Meylan *et al.* 1995). More recently, green sea turtle nesting occurred on Bald Head Island, NC (just east of the mouth of the Cape Fear River), Onslow Island, NC and Cape Hatteras National Seashore. One green sea turtle nested on a beach in Delaware in 2011, although its occurrence was considered very rare.

<u>Threats</u>

Green sea turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green sea turtles appear to be particularly susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles appear to have the highest incidence of disease and the most extensive lesions, whereas lesions in nesting adults are rare. Also, green sea turtles frequenting nearshore waters, areas adjacent to large human populations, and lagoons, areas with low water turnover, 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, and may cause death (George 1997).

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

Atlantic trawl and gillnet fisheries. Murray (2009a) also lists five observed captures of green turtles in Mid-Atlantic sink gillnet gear between 1995 and 2006.

Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer-reviewed publications and NMFS documents (*e.g.*, Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The 2012 section 7 consultation on the shrimp fishery was unable to estimate the total annual level of take for green sea turtles. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least hundreds and possibly low thousands of interactions with green sea turtles annually, of which hundreds are expected to be lethal (NMFS 2012a).

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

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

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

*Summary of Status of Green Sea Turtles*
A review of 32 Index Sites[26] distributed globally revealed a 48-67% decline in the number of mature females nesting annually over the last three generations[27] (Seminoff 2004). An evaluation of green sea turtle nesting sites was also conducted as part of the five-year status review of the species (NMFS and USFWS 2007d). Of the 23 threatened nesting groups assessed in that report for which nesting abundance trends could be determined, ten were considered to be increasing, nine were considered stable, and four were considered to be decreasing (NMFS and USFWS 2007d). Nesting groups were considered to be doing relatively well (the number of sites with increasing nesting were greater than the number of sites with decreasing nesting) in the Pacific, western Atlantic, and central Atlantic (NMFS and USFWS 2007d). However, nesting populations were determined to be doing relatively poorly in Southeast Asia, eastern Indian Ocean, and the Mediterranean. Overall, based on mean annual reproductive effort, the report estimated that 108,761 to 150,521 females nest each year among the 46 threatened and endangered nesting sites included in the evaluation (NMFS and USFWS 2007d). However, given the late age of maturity for green sea turtles, caution is urged regarding the status for any of the nesting groups, since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

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

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

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

[26] 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.
[27] Generation times ranged from 35.5 years to 49.5 years for the assessment depending on the Index Beach site

should be identified (NMFS and USFWS 2007d). Based on the information presented above, for purposes of this Opinion, we consider that the status of green sea turtles over the next ten years will be no worse than it is currently and that the status of the species in the Atlantic Ocean may actually be stable or improving due to increased nesting.

## 4.4 Status of Atlantic Sturgeon

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

The Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a subspecies of sturgeon distributed along the eastern coast of North America from Hamilton Inlet, Labrador, Canada to Cape Canaveral, FL (Scott and Scott 1988; ASSRT 2007;). NMFS has divided U.S. populations of Atlantic sturgeon into five DPSs[28] (77 FR 5880 and 77 FR 5914). These are: the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs (see Figure 1).

The results of genetic studies suggest that natal origin influences the distribution of Atlantic sturgeon in the marine environment (Wirgin and King 2011). However, genetic data, as well as tracking and tagging data, demonstrate that sturgeon from each DPS and Canada occur throughout the full range of the subspecies. Therefore, sturgeon originating from any of the five DPSs can be affected by threats in the marine, estuarine, and riverine environment that occur far from natal spawning rivers.

On February 6, 2012, we published notice in the *Federal Register* that we were listing the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs as "endangered," and the Gulf of Maine DPS as "threatened" (77 FR 5880 and 77 FR 5914). The effective date of the listings is April 6, 2012. The DPSs do not include Atlantic sturgeon spawned in Canadian rivers. Therefore, fish that originated in Canada are not included in the listings. As described below, individuals originating from all five listed DPSs may occur in the action area. Information general to all Atlantic sturgeon, as well as information specific to each of the DPSs, is provided below.

### *Atlantic Sturgeon Life History*

Atlantic sturgeon are long-lived (approximately 60 years), late maturing, estuarine

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

dependent, anadromous[29] fish (Bigelow and Schroeder 1953; Vladykov and Greeley 1963; Mangin 1964; Pikitch *et al.* 2005; Dadswell 2006; ASSRT 2007). They are a relatively large fish, even among sturgeon species (Pikitch *et al.* 2005) and can grow to over 14 feet weighing 800 pounds. Atlantic sturgeon are bottom feeders that suck food into a ventral protruding mouth (Bigelow and Schroeder 1953). Four barbels in front of the mouth assist the sturgeon in locating prey (Bigelow and Schroeder 1953). Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007).

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

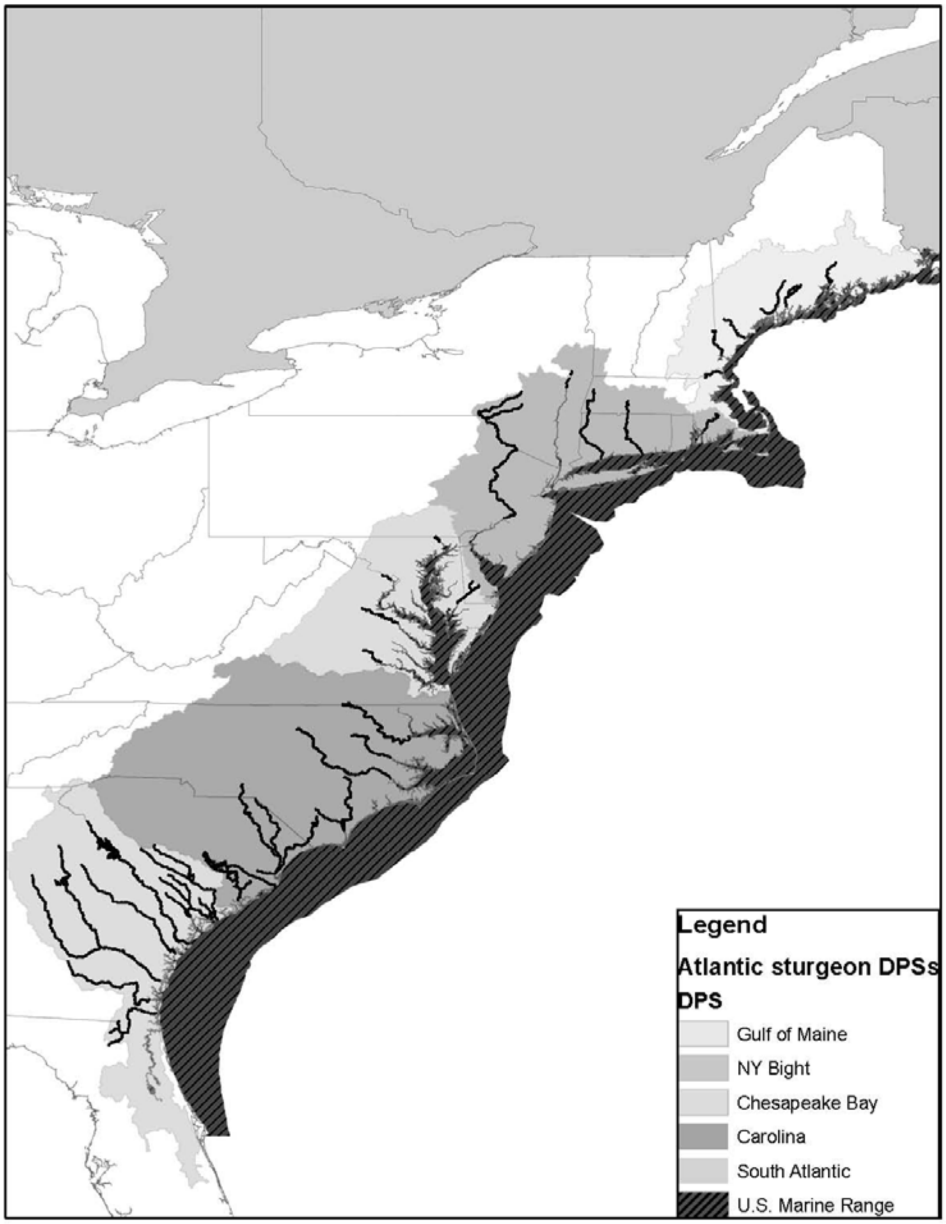


**Figure 1** Geographic Locations for the Five ESA-listed DPSs of Atlantic Sturgeon

Rate of maturation is affected by water temperature and gender. In general: (1) Atlantic sturgeon that originate from southern systems grow faster and mature sooner than Atlantic sturgeon that originate from more northern systems; (2) males grow faster than females; (3) fully mature females attain a larger size (i.e. length) than fully mature males. The largest recorded Atlantic sturgeon was a female captured in 1924 that measured approximately 4.26 meters (Vladykov and Greeley 1963). Dadswell (2006) reported seeing seven fish of comparable size in the St. John River estuary from 1973 to 1995. Observations of large-sized sturgeon are particularly important given that egg production is correlated with age and body size (Smith *et al.* 1982; Van Eenennaam *et al.* 1996; Van Eenennaam and Doroshov 1998; Dadswell 2006). The lengths of Atlantic sturgeon caught since the mid-late 20th century have typically been less than three meters (Smith *et al.* 1982; Smith and Dingley 1984; Smith 1985; Scott and Scott 1988; Young *et al.* 1998; Collins *et al.* 2000; Caron *et al.* 2002; Dadswell 2006; ASSRT 2007; Kahnle *et al.* 2007; DFO, 2011). While females are prolific, with egg production ranging from 400,000 to 4 million eggs per spawning year, females spawn at intervals of two to five years (Vladykov and Greeley 1963; Smith *et al.*, 1982; Van Eenennaam *et al.* 1996; Van Eenennaam and Doroshov 1998; Stevenson and Secor 1999; Dadswell 2006). Given spawning periodicity and a female's relatively late age to maturity, the age at which 50% of the maximum lifetime egg production is achieved is estimated to be 29 years (Boreman 1997). Males exhibit spawning periodicity of one to five years (Smith 1985; Collins *et al.* 2000; Caron *et al.* 2002). While long-lived, Atlantic sturgeon are exposed to a multitude of threats prior to achieving maturation and have a limited number of spawning opportunities once mature.

Water temperature plays a primary role in triggering the timing of spawning migrations (ASMFC, 2009). Spawning migrations generally occur during February-March in southern systems, April-May in Mid-Atlantic systems, and May-July in Canadian systems (Murawski and Pacheco 1977; Smith 1985; Bain 1997; Smith and Clugston 1997; Caron *et al.* 2002). Male sturgeon begin upstream spawning migrations when waters reach approximately 6°C (43° F) (Smith *et al.* 1982; Dovel and Berggren 1983; Smith 1985; ASMFC 2009), and remain on the spawning grounds throughout the spawning season (Bain 1997). Females begin spawning migrations when temperatures are closer to 12°to 13°C (54° to 55°F) (Dovel and Berggren 1983; Smith 1985; Collins *et al.* 2000), make rapid spawning migrations upstream, and quickly depart following spawning (Bain 1997).

The spawning areas in most U.S. rivers have not been well defined. However, the habitat characteristics of spawning areas have been identified based on historical accounts of where fisheries occurred, tracking and tagging studies of spawning sturgeon, and physiological needs of early life stages. Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers, when and where optimal flows are 46-76 centimeters per second and depths are 3-27 meters (Borodin 1925; Dees 1961; Leland 1968; Scott and Crossman 1973; Crance 1987; Shirey *et al.* 1999; Bain *et al.* 2000; Collins *et al.* 2000; Caron *et al.* 2002; Hatin *et al.* 2002; ASMFC 2009). Sturgeon eggs are deposited on hard bottom substrate such as cobble, coarse sand, and bedrock (Dees 1961; Scott and Crossman 1973; Gilbert 1989; Smith and Clugston 1997; Bain *et al.* 2000; Collins *et al.* 2000; Caron *et al.* 2002; Hatin *et al.* 2002; Mohler 2003; ASMFC 2009), and become adhesive shortly after fertilization (Murawski and Pacheco 1977; Van den Avyle 1984; Mohler 2003). Incubation time for the eggs increases as water temperature decreases (Mohler 2003). At temperatures of 20° and 18° C, hatching occurs approximately 94 and 140 hours, respectively, after egg deposition (ASSRT 2007).

Larval Atlantic sturgeon (i.e. less than four weeks old, with total lengths (TL) less than 30 millimeters; Van Eenennaam *et al.* 1996) are assumed to mostly live on or near the bottom and inhabit the same riverine or estuarine areas where they were spawned (Smith *et al.* 1980; Bain *et al.* 2000; Kynard and Horgan 2002; ASMFC 2009). Studies suggest that age-0 (i.e., young-of-year), age-1, and age-2 juvenile Atlantic sturgeon occur in low salinity waters of the natal estuary (Haley 1999; Hatin *et al.* 2007; McCord *et al.* 2007; Munro *et al.* 2007) while older fish are more salt-tolerant and occur in both high salinity and low salinity waters (Collins *et al.* 2000). Atlantic sturgeon remain in the natal estuary for months to years before emigrating to open ocean as subadults (Holland and Yelverton 1973; Dovel and Berggen 1983; Waldman *et al.* 1996; Dadswell 2006; ASSRT 2007).

After emigration from the natal estuary, subadults and adults travel within the marine environment, typically in waters less than 50 meters in depth, using coastal bays, sounds, and ocean waters (Vladykov and Greeley 1963; Murawski and Pacheco 1977; Dovel and Berggren 1983; Smith 1985; Collins and Smith 1997; Welsh *et al.* 2002; Savoy and Pacileo 2003; Stein *et al.* 2004a; Laney *et al.* 2007; Dunton *et al.* 2010; Erickson *et al.* 2011; Wirgin and King 2011). Tracking and tagging studies reveal seasonal movements of Atlantic sturgeon along the coast. Satellite-tagged adult sturgeon from the Hudson River concentrated in the southern part of the Mid-Atlantic Bight at depths greater than 20 meters during winter and spring, and in the northern portion of the Mid-Atlantic Bight at depths less than 20 meters in summer and fall (Erickson *et al.* 2011). Shirey (Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC 2009) found a similar movement pattern for juvenile Atlantic sturgeon based on recaptures of fish originally tagged in the Delaware River. After leaving the Delaware River estuary during the fall, juvenile Atlantic sturgeon were recaptured by commercial fishermen in nearshore waters along the Atlantic coast as far south as Cape Hatteras, NC from November through early March. In the spring, a portion of the tagged fish re-entered the Delaware River estuary. However, many fish continued a northerly coastal migration through the Mid-Atlantic as well as into southern New England waters, where they were recovered throughout the summer months. Movements as far north as Maine were documented. A southerly coastal migration was apparent from tag returns reported in the fall, with the majority of these tag returns from relatively shallow nearshore fisheries, with few fish reported from waters in excess of 25 meters (C. Shirey, Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC 2009). Areas where migratory Atlantic sturgeon commonly aggregate include the Bay of Fundy (e.g., Minas and Cumberland Basins), Massachusetts Bay, Connecticut River estuary, Long Island Sound, New York Bight, Delaware Bay, Chesapeake Bay, and waters off of North Carolina from the Virginia/North Carolina border to Cape Hatteras at depths up to 24 meters (Dovel and Berggren 1983; Dadswell *et al.* 1984; Johnson *et al.* 1997; Rochard *et al.* 1997; Kynard *et al.* 2000; Eyler *et al.* 2004; Stein *et al.* 2004a; Wehrell 2005; Dadswell 2006; ASSRT 2007; Laney *et al.* 2007). These sites may be used as foraging sites and/or thermal refuge.

*Determination of DPS Composition in the Action Area*
As explained above, the range of all five DPSs overlaps and extends from Canada through Cape Canaveral, FL. We have considered the best available information to determine from which DPSs individuals in the action area are likely to have originated. We have determined that Atlantic sturgeon in the action area likely originate from all five DPSs at the following frequencies: Gulf of Maine (GOM) 11%; New York Bight (NYB) 51%; Chesapeake Bay (CB) 13%; Carolina 2%, and South Atlantic (SA) 22%. Approximately 1% of the Atlantic sturgeon in the action area originate from Canada. These percentages are based on genetic sampling of individuals (n=173) captured during observed fishing trips along the Atlantic coast from Maine through North Carolina, and the results of the genetic analyses for these 173 fish were compared against a reference population of 411 fish and results for an additional 790 fish from other sampling efforts. Therefore, they represent the best available information on the likely genetic makeup of individuals occurring in the action area. The genetic assignments have a plus/minus 5% confidence interval; however, For purposes of section 7 consultation we have selected the reported values without their associated confidence intervals. The reported values, which approximate the mid-point of the range, are a reasonable indication of the likely genetic makeup of Atlantic sturgeon in the action area. These assignments and the data from which they are derived are described in detail in Damon-Randall *et al.* (2013).

*Distribution and Abundance*
Atlantic sturgeon underwent significant range-wide declines from historical abundance levels due to overfishing in the mid to late 19$^{th}$ century when a caviar market was established (Scott and Crossman 1973; Taub 1990; Kennebec River Resource Management Plan 1993; Smith and Clugston 1997; Dadswell 2006; ASSRT 2007). Abundance of spawning-aged females prior to this period of exploitation was predicted to be greater than 100,000 for the Delaware River, and at least 10,000 females for other spawning stocks (Secor and Waldman 1999; Secor 2002). Historical records suggest that Atlantic sturgeon spawned in at least 35 rivers prior to this period. Currently, only 17 U.S. rivers are known to support spawning (i.e., presence of young-of-year or gravid Atlantic sturgeon documented within the past 15 years) (ASSRT 2007). While there may be other rivers supporting spawning for which definitive evidence has not been obtained (e.g., in the Penobscot and York Rivers), the number of rivers supporting spawning of Atlantic sturgeon are approximately half of what they were historically. In addition, only five rivers (Kennebec, Androscoggin, Hudson, Delaware, James) are known to currently support spawning from Maine through Virginia, where historical records show that there used to be 15 spawning rivers (ASSRT 2007). Thus, there are substantial gaps between Atlantic sturgeon spawning rivers among northern and Mid-Atlantic states which could make recolonization of extirpated populations more difficult.

At the time of the listing, there were no current, published population abundance estimates for any of the currently known spawning stocks or for any of the five

DPSs of Atlantic sturgeon. An estimate of 863 mature adults per year (596 males and 267 females) was calculated for the Hudson River based on fishery-dependent data collected from 1985 to 1995 (Kahnle *et al.*, 2007). An estimate of 343 spawning adults per year is available for the Altamaha River, GA, based on fishery-independent data collected in 2004 and 2005 (Schueller and Peterson 2006). Using the data collected from the Hudson and Altamaha Rivers to estimate the total number of Atlantic sturgeon in either subpopulation is not possible, since mature Atlantic sturgeon may not spawn every year (Vladykov and Greeley 1963; Smith 1985; Van Eenennaam *et al.* 1996; Stevenson and Secor 1999; Collins *et al.* 2000; Caron *et al.* 2002), the age structure of these populations is not well understood, and stage-to-stage survival is unknown. In other words, the information that would allow us to take an estimate of annual spawning adults and expand that estimate to an estimate of the total number of individuals (*e.g.*, yearlings, subadults, and adults) in a population is lacking. The ASSRT presumed that the Hudson and Altamaha rivers had the most robust of the remaining U.S. Atlantic sturgeon spawning populations and concluded that the other U.S. spawning populations were likely less than 300 spawning adults per year (ASSRT 2007).

Lacking complete estimates of population abundance across the distribution of Atlantic sturgeon, the NEFSC developed a virtual population analysis model with the goal of estimating bounds of Atlantic sturgeon ocean abundance (see Kocik et al. 2013). The NEFSC suggested that cumulative annual estimates of surviving fishery discards could provide a minimum estimate of abundance. The objectives of producing the Atlantic Sturgeon Production Index (ASPI) were to characterize uncertainty in abundance estimates arising from multiple sources of observation and process error and to complement future efforts to conduct a more comprehensive stock assessment (Table 12). The ASPI provides a general abundance metric to assess risk for actions that may affect Atlantic sturgeon in the ocean; however, it is not a comprehensive stock assessment. In general, the model uses empirical estimates of post-capture survivors and natural survival, as well as probability estimates of recapture using tagging data from the United States Fish and Wildlife Service (USFWS) sturgeon tagging database, and federal fishery discard estimates from 2006 to 2010 to produce a virtual population. The USFWS sturgeon tagging database is a repository for sturgeon tagging information on the Atlantic coast. The database contains tag, release, and recapture information from state and federal researchers. The database records recaptures by the fishing fleet, researchers, and researchers on fishery vessels.

In additional to the ASPI, a population estimate was derived from the Northeast Area Monitoring and Assessment Program (NEAMAP) (Table 12). NEAMAP trawl surveys are conducted from Cape Cod, Massachusetts to Cape Hatteras, North Carolina in nearshore waters at depths up to 18.3 meters (60 feet) during the fall since 2007 and spring since 2008. Each survey employs a spatially stratified random design with a total of 35 strata and 150 stations. The ASMFC has initiated a new stock assessment with the goal of completing it by the end of 2014. NOAA Fisheries will be partnering with them to conduct the stock assessment, and the ocean population abundance estimates produced by the NEFSC will be shared with the stock assessment committee for consideration in the stock assessment.

**Table 12** Description of the ASPI model and NEAMAP survey based area estimate method.

| Model Name | Model Description |
| --- | --- |
| A. ASPI | Uses tag-based estimates of recapture probabilities from 1999 to 2009. Natural mortality based on Kahnle *et al.* (2007) rather than estimates derived from tagging model. Tag recaptures from commercial fisheries are adjusted for non reporting based on recaptures from observers and researchers. Tag loss assumed to be zero. |
| B. NEAMAP Swept Area | Uses NEAMAP survey-based swept area estimates of abundance and assumed estimates of gear efficiency. Estimates based on average of ten surveys from fall 2007 to spring 2012. |

**Table 13**: **Modeled Results**

| Model Run | Model Years | 95% low | Mean | 95% high |
| --- | --- | --- | --- | --- |
| A. ASPI | 1999-2009 | 165,381 | 417,934 | 744,597 |
| B.1 NEAMAP Survey, swept area assuming 100% efficiency | 2007-2012 | 8,921 | 33,888 | 58,856 |
| B.2 NEAMAP Survey, swept area assuming 50% efficiency | 2007-2012 | 13,962 | 67,776 | 105,984 |
| B.3 NEAMAP Survey, swept area assuming 10% efficiency | 2007-2012 | 89,206 | 338,882 | 588,558 |

As illustrated by Table 13 above, the ASPI model projects a mean population size of 417,934 Atlantic sturgeon and the NEAMAP Survey projects mean population sizes ranging from 33,888 to 338,882 depending on the assumption made regarding efficiency of that survey. As noted above, the ASPI model uses empirical estimates of post-capture survivors and natural survival, as well as probability estimates of recapture using tagging data from the United States Fish and Wildlife Service (USFWS) sturgeon tagging database, and federal fishery discard estimates from 2006 to 2010 to produce a virtual population. The NEAMAP estimate, in contrast, is more empirically derived and does not depend on as many assumptions. For the purposes of this Opinion, while the ASPI model is considered as part of the ASMFC stock assessment, we consider the NEAMAP estimate as the best available information on population size.

Once we have selected the NEAMAP method, we must then determine the most appropriate estimate of the efficiency of that survey. Atlantic sturgeon are frequently encountered during the NEAMAP surveys. The information from this survey can be used to calculate minimum swept area population estimates within the strata swept by the survey. The estimate from fall surveys ranges from 6,980 to 42,160 with coefficients of variation between 0.02 and 0.57, and the estimates from spring surveys ranges from 25,540 to 52,990 with coefficients of variation between 0.27 and 0.65 (Table 14). These are considered minimum estimates because the calculation makes the assumption that the gear will capture (i.e. net efficiency) 100% of the sturgeon in the water column along the tow path and that all sturgeon are with the sampling domain of the survey. We define catchability as 1) the product of the probability of capture given encounter (i.e. net efficiency), and 2) the fraction of the population within the sampling domain. Catchabilities less than 100% will result in estimates greater than the minimum. The true catchability depends on many factors including the availability of the species to the survey and the behavior of the species with respect to the gear. True catchabilities much less than 100% are common for most species. The ratio of total sturgeon habitat to area sampled by the NEAMAP survey is unknown, but is certainly greater than one (i.e. the NEAMAP survey does not survey 100% of the Atlantic sturgeon habitat).

**Table 14** Annual minimum swept area estimates for Atlantic sturgeon during the spring and fall from the Northeast Area Monitoring and Assessment Program survey. Estimates assume 100% net efficiencies. Estimates provided by Dr. Chris Bonzek, Virginia Institute of Marine Science (VIMS).

| Year | Fall Number | CV | Spring Number | CV |
|---|---|---|---|---|
| 2007 | 6,981 | 0.015 | | |
| 2008 | 33,949 | 0.322 | 25,541 | 0.391 |
| 2009 | 32,227 | 0.316 | 41,196 | 0.353 |
| 2010 | 42,164 | 0.566 | 52,992 | 0.265 |
| 2011 | 22,932 | 0.399 | 52,840 | 0.480 |
| 2012 | | | 28,060 | 0.652 |

Available data do not support estimation of true catchabilty (i.e., net efficiency X availability) of the NEAMAP trawl survey for Atlantic sturgeon. Thus, the NEAMAP swept area biomass estimates were produced and presented in Kocik et al. (2013) for catchabilities from 5 to 100%. In estimating the efficiency of the sampling net, we consider the likelihood that an Atlantic sturgeon in the survey area is likely to be captured by the trawl. True efficiencies less than 100% are common for most species. Assuming the NEAMAP surveys have been 100% efficient would require the unlikely assumption that the survey gear captures all Atlantic sturgeon within the path of the trawl and all sturgeon are within the sampling area of the NEAMAP survey. In estimating the fraction of the Atlantic sturgeon population within the sampling area of the NEAMAP, we consider that the NEAMAP-based estimates do not include young of the year fish and juveniles in the rivers; however, those segments of the Atlantic sturgeon populations are not at risk from federal commercial fisheries considered as part of the proposed action since they do not occur within the action area. Additionally, although the NEAMAP surveys are not conducted in the Gulf of Maine or south of Cape Hatteras, NC, the NEAMAP surveys are conducted throughout the majority of the action area from Cape Cod to Cape Hatteras at depths up to 18.3 meters (60 feet), which includes the preferred depth ranges of subadult and adult Atlantic sturgeon. NEAMAP surveys take place during seasons that coincide with known Atlantic sturgeon coastal migration patterns in the ocean. Therefore, the NEAMAP estimates are minimum estimates of the ocean population of Atlantic sturgeon but are based on sampling in much of the action area, in known sturgeon coastal migration areas during times that sturgeon are expected to be migrating north and south.

Based on the above, we consider that the NEAMAP samples an area utilized by Atlantic sturgeon, but does not sample all the locations and times where Atlantic sturgeon are present and the trawl net captures some, but likely not all, of the Atlantic sturgeon present in the sampling area. Therefore, we assumed that net efficiency and the fraction of the population exposed to the NEAMAP survey in combination result in a 50% catchability. The 50% catchability assumption seems to reasonably account for the robust, yet not complete sampling of the Atlantic sturgeon oceanic temporal and spatial ranges and the documented high rates of encounter with NEAMAP survey gear and Atlantic sturgeon. For this Opinion, we have determined that the best available data at this time are the population estimates derived from NEAMAP swept area biomass resulting from the 50% catchability rate.

The ocean population abundance of 67,776 fish estimated from the NEAMAP survey assuming 50% efficiency was subsequently partitioned by DPS based on genetic frequencies of occurrence (Table 15). Given the proportion of adults to subadults in the observer database (approximate ratio of 1:3), we have also estimated a number of subadults originating from each DPS. However, this cannot be considered an estimate of the total number of subadults because it only considers those subadults that are of a size vulnerable to capture in commercial sink gillnet and otter trawl gear in the marine environment and are present in the marine environment.

**Table 15** Summary of calculated population estimates based upon the NEAMAP Survey swept area assuming 50% efficiency

| DPS | Estimated Ocean Population Abundance | Estimated Ocean Population of Adults | Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries) |
|---|---|---|---|
| **GOM (11%)** | 7,455 | 1,864 | 5,591 |

| | | | |
|---|---|---|---|
| **NYB (51%)** | 34,566 | 8,642 | 25,925 |
| **CB (13%)** | 8,811 | 2,203 | 6,608 |
| **Carolina (2%)** | 1,356 | 339 | 1,017 |
| **SA (22%)** | 14,911 | 3,728 | 11,183 |
| **Canada (1%)** | 678 | 170 | 509 |

*Threats Faced by Atlantic Sturgeon Throughout Their Range*
Atlantic sturgeon are susceptible to over-exploitation given their life history characteristics (e.g., late maturity and dependence on a wide variety of habitats). Similar to other sturgeon species (Vladykov and Greeley 1963; Pikitch *et al.* 2005), Atlantic sturgeon experienced range-wide declines from historical abundance levels due to overfishing (for caviar and meat) and impacts to habitat in the 19th and 20th centuries (Taub 1990; Smith and Clugston 1997; Secor and Waldman 1999).

Because a DPS is a group of populations, the stability, viability, and persistence of individual populations affects the persistence and viability of the larger DPS. The loss of any population within a DPS could result in: (1) a long-term gap in the range of the DPS that is unlikely to be recolonized; (2) loss of reproducing individuals; (3) loss of genetic biodiversity; (4) loss of unique haplotypes; (5) loss of adaptive traits; and (6) reduction in total number. The loss of a population will negatively impact the persistence and viability of the DPS as a whole, as fewer than two individuals per generation spawn outside their natal rivers (Secor and Waldman 1999). The persistence of individual populations, and in turn the DPS, depends on successful spawning and rearing within the freshwater habitat, emigration to marine habitats to grow, and return of adults to natal rivers to spawn.

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

Atlantic sturgeon are particularly sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum fecundity values, and a large percentage of egg production occurs later in life. Based on these life history traits, Boreman (1997) calculated that Atlantic sturgeon can only withstand the annual loss of up to 5% of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0 and 51%, with the greatest mortality occurring in sturgeon caught by sink gillnets. Atlantic sturgeon are particularly vulnerable to being caught in sink gillnets; therefore, fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch. Fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

As a wide-ranging anadromous species, Atlantic sturgeon are subject to numerous federal (U.S. and Canadian), state and provincial, and inter-jurisdictional laws, regulations, and agency activities. While these mechanisms, including the prohibition on possession, have addressed impacts to Atlantic sturgeon through directed fisheries, the listing determination concluded that the mechanisms in place to address the risk posed to Atlantic sturgeon from commercial bycatch were insufficient.

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

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

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

Bycatch in U.S. waters is one of the primary threats faced by all five DPSs. At this time, we have an estimate of the number of Atlantic sturgeon captured and killed in sink gillnet and otter trawl fisheries authorized by federal FMPs (NMFS NEFSC 2011b) in the Northeast Region but do not have a similar estimate for southeast fisheries. We also do not have an estimate of the number of Atlantic sturgeon captured or killed in state fisheries. At this time, we are not able to quantify the effects of other significant threats (e.g., vessel strikes, poor water quality, water availability, dams, and dredging) in terms of habitat impacts or loss of individuals. While we have some information on the number of mortalities that have occurred in the past in association with certain activities (e.g., mortalities in the Delaware and James Rivers that are thought to be due to vessel strikes), we are not able to use those numbers to extrapolate effects throughout one or more DPSs. This is because of (1) the small number of data points and, (2) the lack of information on the percent of incidents that the observed mortalities represent.

As noted above, the NEFSC prepared an estimate of the number of encounters of Atlantic sturgeon in fisheries authorized by Northeast FMPs (NMFS NEFSC 2011b). The analysis estimates that from 2006 through 2010, there were averages of 1,548 and 1,569 encounters per year in observed gillnet and trawl fisheries, respectively, with an average of 3,118 encounters combined annually. Mortality rates in gillnet gear were approximately 20%. Mortality rates in otter trawl gear are generally lower, at approximately 5%.

Global climate change may affect all DPSs of Atlantic sturgeon in the future; however, effects of increased water temperature and decreased water availability are most likely to affect the South Atlantic and Carolina DPSs. Implications of climate change to the Atlantic sturgeon DPSs have been speculated, yet no scientific data are available on past trends related to climate effects on this species, and current scientific methods are not able to reliably predict the future magnitude of climate change and associated impacts or the adaptive capacity of these species. Impacts of climate change on Atlantic sturgeon are uncertain at this time, and cannot be quantified. Any prediction of effects is made more difficult by a lack of information on the rate of expected change in conditions and a lack of information on the adaptive capacity of the species (i.e., its ability to evolve to cope with a changing environment). For analysis on the potential effects of climate change on Atlantic sturgeon, see Section 6.2.3 below.

#### 4.4.1 Status of Gulf of Maine DPS

The GOM DPS includes the following: all anadromous Atlantic sturgeon that spawn or are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the GOM as far south as Chatham, MA. The marine range of Atlantic sturgeon from the GOM DPS extends from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the GOM DPS and the adjacent portion of the marine range are shown in Figure

1. Within this range, Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (ASSRT 2007). Spawning still occurs in the Kennebec and Androscoggin Rivers, and it is possible that it still occurs in the Penobscot River as well. Spawning in the Androscoggin River was just recently confirmed by the Maine Department of Marine Resources when they captured a larval Atlantic sturgeon during the 2011 spawning season below the Brunswick Dam. There is no evidence of recent spawning in the remaining rivers. In the 1800s, construction of the Essex Dam on the Merrimack River at river kilometer (rkm) 49 blocked access to 58% of Atlantic sturgeon habitat in the river (Oakley 2003; ASSRT 2007). However, the accessible portions of the Merrimack seem to be suitable habitat for Atlantic sturgeon spawning and rearing (i.e., nursery habitat) (Keiffer and Kynard 1993). Therefore, the availability of spawning habitat does not appear to be the reason for the lack of observed spawning in the Merrimack River. Studies are ongoing to determine whether Atlantic sturgeon are spawning in the Penobscot and Saco Rivers. Atlantic sturgeon that are spawned elsewhere continue to use habitats within these rivers as part of their overall marine range (ASSRT 2007).

At its mouth, the Kennebec River drains an area of 24,667 square kilometers, and is part of a large estuarine system that includes the Androscoggin and Sheepscot Rivers (ASMFC 1998a; NMFS and USFWS 1998d; Squiers 1998). The Kennebec and Androscoggin Rivers flow into Merrymeeting Bay, a tidal freshwater bay, and exit as a combined river system through a narrow channel, flowing approximately 32 kilometers (20 miles) to the Atlantic Ocean as the tidal segment of the Kennebec River (Squiers 1998). This lower tidal segment of the Kennebec River forms a complex with the Sheepscot River estuary (ASMFC 1998a; Squiers 1998).

Substrate type in the Kennebec estuary is largely sand and bedrock (Fenster and Fitzgerald 1996; Moore and Reblin 2010). Main channel depths at low tide typically range from 17 meters (58 feet) near the mouth to less than 10 meters (33 feet) in the Kennebec River above Merrymeeting Bay (Moore and Reblin 2010). Salinities range from 31 parts per thousand at Parker Head (5 kilometers from the mouth) to 18 parts per thousand at Doubling Point during summer low flows (ASMFC 1998a). The 14-kilometer river segment above Doubling Point to Chops Point (the outlet of Merrymeeting Bay) is an area of transition (mid estuary) (ASMFC 1998a). The salinities in this section vary both seasonally and over a tidal cycle. During spring freshets this section is entirely fresh water but during summer low flows, salinities can range from 2 to 3 parts per thousand at Chops Point to 18 parts per thousand at Doubling Point (ASMFC 1998a). The river is essentially tidal freshwater from the outlet of Merrymeeting Bay upriver to the site of the former Edwards Dam (ASMFC 1998a). Mean tidal amplitude ranges from 2.56 meters at the mouth of the Kennebec River estuary to 1.25 meters in Augusta near the head of tide on the Kennebec River (in the vicinity of the former Edwards Dam) and 1.16 meters at Brunswick on the Androscoggin River (ASMFC 1998a).

Bigelow and Schroeder (1953) surmised that Atlantic sturgeon likely spawned in Gulf of Maine Rivers in May-July. More recent captures of Atlantic sturgeon in spawning condition within the Kennebec River suggest that spawning more likely occurs in June-July (Squiers *et al.* 1981; ASMFC 1998a; NMFS and USFWS 1998d). Evidence for the timing and location of Atlantic sturgeon spawning in the Kennebec River includes: (1) the capture of five adult male Atlantic sturgeon in spawning condition (i.e., expressing milt) in July 1994 below the (former) Edwards Dam; (2) capture of 31 adult Atlantic sturgeon from June 15 through July 26,1980 in a small commercial fishery directed at Atlantic sturgeon from the South Gardiner area (above Merrymeeting Bay) that included at least four ripe males and one ripe female captured on July 26,1980; and, (3) capture of nine adults during a gillnet survey conducted from 1977 to 1981, the majority of which were captured in July in the area from Merrymeeting Bay and upriver as far as Gardiner, ME (NMFS and USFWS 1998d; ASMFC 2007). The low salinity of waters above Merrymeeting Bay are consistent with values found in other rivers where successful Atlantic sturgeon spawning is known to occur.

Age to maturity for GOM DPS Atlantic sturgeon is unknown. However, Atlantic sturgeon riverine populations exhibit clinal variation with faster growth and earlier age to maturity for those that originate from southern waters, and slower growth and later age to maturity for those that originate from northern waters (75 FR 61872; October 6, 2010). Age at maturity is 11 to 21 years for Atlantic sturgeon originating from the Hudson River (Young *et al.* 1998), and 22 to 34 years for Atlantic sturgeon that originate from the Saint Lawrence River (Scott and Crossman1973). Therefore, age at maturity for Atlantic sturgeon of the GOM DPS likely falls within these values. Of the 18 sturgeon examined from the commercial fishery that occurred in the Kennebec River in 1980, all of which were considered mature, age estimates for the 15 males ranged from 17-40 years, and from 25-40 years old for the three females (Squiers *et al.* 1981).

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

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

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

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

There are no direct in-river abundance estimates for the GOM DPS. The Atlantic Sturgeon Status Review Team (ASSRT) (2007) presumed that the GOM DPS was comprised of less than 300 spawning adults per year, based on extrapolated abundance estimates from the Hudson and Altamaha riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over two time periods, 1977-1981 and 1998-2000, resulted in the capture of nine adult Atlantic sturgeon (Squiers 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the capture gear used may not have been selective for the larger-sized adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies. As described earlier in Section 4.4, we have estimated that there are a minimum of 7,455 GOM DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries. We note further that this estimate is predicated on the assumption that fish in the GOM DPS would be available for capture in the NEAMAP survey which extends from Block Island Sound (RI) southward.

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

Some of the impacts from the threats that contributed to the decline of the GOM DPS have been removed (*e.g.*, directed fishing), or reduced as a result of improvements in water quality and removal of dams (*e.g.*, the Edwards Dam on the Kennebec River in 1999). In Maine state waters, there are strict regulations on the use of fishing gear that incidentally catches sturgeon. In addition, in the last several years there have been reductions in fishing effort in state and federal waters, which most likely would result in a reduction in bycatch mortality of Atlantic sturgeon. A significant amount of fishing in the Gulf of Maine is conducted using trawl gear, which is known to have a much lower mortality rate for Atlantic sturgeon caught in the gear compared to sink gillnet gear (ASMFC 2007). Atlantic sturgeon from the GOM DPS are not commonly taken as bycatch in areas south of Chatham, MA, with only 8% (e.g., 7 of 84 fish) of interactions observed south of Chatham being assigned to the GOM DPS (Wirgin and King 2011). Tagging results also indicate that GOM DPS fish tend to remain within the waters of the Gulf of Maine and only occasionally venture to points south.

Data on Atlantic sturgeon incidentally caught in trawls and intertidal fish weirs fished in the Minas Basin area of the Bay of Fundy (Canada) indicate that approximately 35 % originated from the GOM DPS (Wirgin *et al.* 2012). Thus, a significant number of the GOM DPS fish appear to migrate north into Canadian waters where they may be subjected to a variety of threats including bycatch.

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

### 4.4.2 Status of New York Bight DPS

The NYB DPS includes the following: all anadromous Atlantic sturgeon that spawn or are spawned in the watersheds that drain into coastal waters from Chatham, MA to the Delaware-Maryland border on Fenwick Island. The marine range of Atlantic sturgeon from the NYB DPS extends from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the NYB DPS and the adjacent portion of the marine range are shown in Figure 1. Within this range, Atlantic sturgeon historically spawned in the Connecticut, Delaware, Hudson, and Taunton Rivers (Murawski and Pacheco 1977; Secor 2002; ASSRT 2007). Spawning still occurs in the Delaware and Hudson Rivers, but there is no recent evidence (within the last 15 years) of spawning in the Connecticut and Taunton Rivers (ASSRT 2007). Atlantic sturgeon that are spawned elsewhere continue to use habitats within the Connecticut and Taunton Rivers as part of their overall marine range (ASSRT 2007; Savoy 2007; Wirgin and King 2011).

The abundance of the Hudson River Atlantic sturgeon riverine population before the over-exploitation of the 1800s is unknown but has been conservatively estimated at 6,000 adult females (Secor 2002). Current abundance is likely at least one order of magnitude smaller than historical levels (Secor 2002; ASSRT 2007; Kahnle *et al.* 2007). As described above, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population based on fishery-dependent data collected from 1985 to 1995 (Kahnle *et al.* 2007). Kahnle *et al.* (1998; 2007) also showed that the level of fishing mortality from the Hudson River Atlantic sturgeon fishery during the period of 1985-1995 exceeded the estimated sustainable level of fishing mortality for the riverine population and may have led to reduced recruitment. All available data on abundance of juvenile Atlantic sturgeon in the Hudson River Estuary indicate a substantial drop in production of young since the mid 1970s (Kahnle *et al.* 1998). A decline appeared to occur in the mid to late 1970's followed by a secondary drop in the late 1980s (Kahnle *et al.* 1998; Sweka *et al.* 2007; ASMFC 2010) CPUE data suggests that recruitment has remained depressed relative to catches of juvenile Atlantic sturgeon in the estuary during the mid-late 1980s (Sweka *et al.* 2007; ASMFC 2010). The CPUE data from 1985 to 2011 show significant fluctuations. There appears to be a decline in the number of juveniles between the late 1980s and early 1990s and then a slight increase in the 2000s, but, given the significant annual fluctuation, it is difficult to discern any real trend. Despite the CPUEs from 2000 to 2011 being slightly higher than those from 1990 to 1999, they are low compared to the mid to late 1980s (Figure 2). There is currently not enough information regarding any life stage to establish a trend for the Hudson River population.

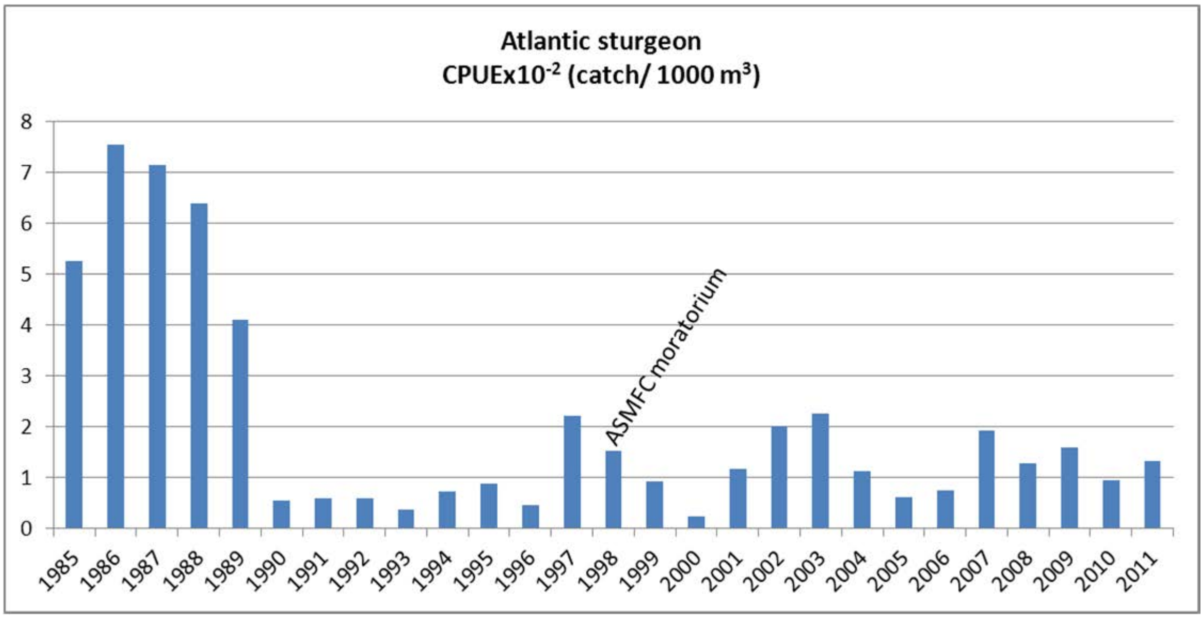


**Figure 2** Hudson River Atlantic sturgeon CPUE juvenile index (1985-present).

There is no overall, empirical abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800s indicate that this was historically a large population with an estimated 180,000 adult females prior to 1890 (Secor and Waldman 1999; Secor 2002). Sampling in 2009 to target young-of-the year (YOY) Atlantic sturgeon in the Delaware River (*i.e.*, natal sturgeon) resulted in the capture of 34 YOY, ranging in size from 178 to 349 millimeters TL (Fisher 2009), and the collection of 32 YOY Atlantic sturgeon in a separate study (Brundage and O’Herron in Calvo *et al.* 2010). Genetics information collected from 33 of these YOY indicates that at least three females successfully contributed to the 2009 year class (Fisher 2011). Therefore, while the capture of YOY in 2009 provides evidence that successful spawning is still occurring in the Delaware River, the relatively low numbers suggest the existing riverine population is small.

Several threats play a role in shaping the current status and trends observed in the Delaware River and Estuary. Mortalities associated with bycatch in fisheries in state and federal waters occur. In the marine range, NYB DPS Atlantic sturgeon are incidentally captured in federal and state-managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004b; ASMFC 2007). As explained above, we have estimates of the number of subadults and adults that are killed as a result of bycatch in fisheries authorized under Northeast FMPs. At this time, we are not able to quantify the impacts from other threats or estimate the number of individuals killed as a result of other anthropogenic threats. In-river threats include habitat disturbance from dredging, and impacts from historical pollution and impaired water quality. A dredged navigation channel extends from Trenton seaward through the tidal river (Brundage and O'Herron 2009), and the river receives significant shipping traffic. Vessel strikes have been identified as a threat in the Delaware River and may be detrimental to the long-term viability of the NYB DPS, as well as other DPSs (Brown and Murphy 2010)

*Summary of the New York Bight DPS*

Atlantic sturgeon originating from the NYB DPS spawn in the Hudson and Delaware Rivers. While genetic testing can differentiate between individuals originating from the Hudson or Delaware River, the available information suggests that the straying rate is relatively high between these rivers. Some of the impact from the threats that contributed to the decline of the NYB DPS have been removed (*e.g.,* directed fishing) or reduced as a result of improvements in water quality since passage of the Clean Water Act (CWA). In addition, there have been reductions in fishing effort in state and federal waters, which may result in a reduction in bycatch mortality of Atlantic sturgeon. Nevertheless, areas with persistent, degraded water quality, habitat impacts from dredging, continued bycatch in state and federally managed fisheries, and vessel strikes remain significant threats to the NYB DPS.

In its marine range, NYB DPS Atlantic sturgeon are incidentally captured in federal and state-managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004a; ASMFC 2007). Based on mixed stock analysis results presented by Wirgin and King (2011), more than 40% of the Atlantic sturgeon bycatch interactions in the Mid Atlantic Bight region were sturgeon from the NYB DPS. Individual-based assignment and mixed stock analysis of samples collected from sturgeon captured in Canadian fisheries in the Bay of Fundy indicated that approximately 1-2% were from the NYB DPS (Wirgin *et al.* 2012). At this time, we are not able to quantify the impacts from threats other than fisheries or estimate the number of individuals killed as a result of other anthropogenic threats.

Riverine habitat may be impacted by dredging and other in-water activities, disturbing spawning habitat and also altering the benthic forage base. Both the Hudson and Delaware Rivers have navigation channels that are maintained by dredging. Dredging is also used to maintain channels in the nearshore marine environment. Dredging outside of federal channels and in-water construction occurs throughout the New York Bight region. While some dredging projects operate with observers to document fish mortalities, many do not. We have reports of one Atlantic sturgeon entrained during hopper dredging operations in Ambrose

Channel, NJ. We recently consulted on two dredging projects: the ACOE Delaware River Federal Navigation Channel deepening project and on the New York and New Jersey Harbor Deepening Project. In both cases, we determined that while the proposed actions may adversely affect Atlantic sturgeon, they were not likely to jeopardize the continued existence of any DPS of Atlantic sturgeon (NMFS 2012i and NMFS 2012j).

In the Hudson and Delaware Rivers, dams do not block access to historical habitat. The Holyoke Dam on the Connecticut River blocks passage past the dam at Holyoke; however, the extent that Atlantic sturgeon would historically have used habitat upstream of Holyoke is unknown. The first dam on the Taunton River may block access to historical spawning habitat. Connectivity also may be disrupted by the presence of dams on several smaller rivers in the New York Bight region. Because no Atlantic sturgeon occur upstream of any hydroelectric projects in the New York Bight region, passage over hydroelectric dams or through hydroelectric turbines is not a source of injury or mortality in this area. The extent to which Atlantic sturgeon are affected by operations of dams in the New York Bight region is currently unknown. Atlantic sturgeon may also be impinged or entrained at power plants in the Hudson and Delaware Rivers, and may be adversely affected by the operation of the power plants, but the power plants have not been found to jeopardize their continued existence.

NYB DPS Atlantic sturgeon may also be affected by degraded water quality. Rivers in the NYB region, including the Hudson and Delaware, have been heavily polluted by industrial and sewer discharges. In general, water quality has improved in the Hudson and Delaware over the past several decades (Lichter *et al.* 2006; EPA 2008). While water quality has improved and most discharges are limited through regulations, it it likely that pollutants persist in the benthic environment. This can be particularly problematic if pollutants are present on spawning and nursery grounds, where developing eggs and larvae are particularly susceptible to exposure to contaminants.

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

Studies have shown that to rebuild, Atlantic sturgeon can only sustain low levels of anthropogenic mortality (Boreman 1997; ASMFC 2007; Kahnle *et al.* 2007; Brown and Murphy 2010). There are no empirical abundance estimates of the number of Atlantic sturgeon in the NYB DPS. As described in Section 4.4, we have estimated that there are a minimum of 34,566 NYB DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries. We have determined that the NYB DPS is currently at risk of extinction due to: (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and (3) the impacts and threats that have and will continue to affect population recovery.

#### 4.4.3 Status of Chesapeake Bay DPS

The CB DPS includes the following: all anadromous Atlantic sturgeons that spawn or are spawned in the watersheds that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, VA. The marine range of Atlantic sturgeon from the CB DPS extends from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the CB DPS and the adjacent portion of the marine range are shown in Figure 1. Within this range, Atlantic sturgeon historically spawned in the Susquehanna, Potomac, James, York, Rappahannock, and Nottoway Rivers (ASSRT 2007). Based on the review by Oakley (2003), 100 % of Atlantic sturgeon habitat is currently accessible in these rivers since most of the barriers to passage (*i.e.* dams) are located upriver of where spawning is expected to have historically occurred (ASSRT 2007). Spawning still occurs in the James River, and the presence of juvenile and adult sturgeon in the York River suggests that spawning may occur there as well (Musick *et al.* 1994; ASSRT 2007; Greene *et al.* 2009). However, conclusive evidence of current spawning is only available for the James River, where a recent study found evidence of Atlantic sturgeon spawning in the fall (Balazik *et al.* 2012). Atlantic sturgeon that are spawned elsewhere are known to use the Chesapeake Bay for other life functions, such as foraging and as juvenile nursery habitat (Vladykov and Greeley 1963; ASSRT 2007; Wirgin *et al.* 2007; Grunwald *et al.* 2008).

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

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

Decreased water quality also threatens Atlantic sturgeon of the Chesapeake Bay DPS, especially since the Chesapeake Bay system is vulnerable to the effects of nutrient enrichment due to a relatively low tidal exchange and flushing rate, large surface-to-volume ratio, and strong stratification during the spring and summer months (Pyzik *et al.* 2004; ASMFC 1998a; ASSRT 2007; EPA 2008). These conditions contribute to reductions in DO levels throughout the Bay. The availability of nursery habitat, in particular, may be limited given the recurrent hypoxic (low DO) conditions within the Bay (Niklitschek and Secor 2005; 2010). Heavy industrial development during the twentieth century in rivers inhabited by sturgeon impaired water quality and impeded these species' recovery.

Although there have been improvements in the some areas of the Bay's health, the ecosystem remains in poor condition. EPA gave the overall health of the Bay a grade of 45% based on goals for water quality, habitats, lower food web productivity, and fish and shellfish abundance (EPA CBP 2010). This was a 6% increase from 2008. According to EPA, the modest gain in the health score was due to a large increase in adult blue crab population, expansion of underwater grass beds growing in the Bay's shallows, and improvements in water clarity and bottom habitat health as highlighted below:

- 12% of the Bay and its tidal tributaries met Clean Water Act standards for DO between 2007 and 2009, a decrease of 5% from 2006-2008.
- 26% of the tidal waters met or exceeded guidelines for water clarity, a 12% increase from 2008.
- Underwater bay grasses covered 9,039 more acres of the Bay's shallow waters for a total of 85,899 acres, 46% of the Bay-wide goal.
- The health of the Bay's bottom dwelling species reach a record high of 56% of the goal, improving by approximately 15 Bay-wide.
- The adult blue crab population increased to 223 million, its highest level since 1993.

At this time we do not have sufficient information to quantify the extent that degraded water quality effects habitat or individuals in the James River or throughout the Chesapeake Bay.

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

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

*Summary of the Chesapeake Bay DPS*
Spawning for the CB DPS is known to occur in only the James River. Spawning may be occurring in other rivers, such as the York, but has not been confirmed. There are anecdotal reports of increased sightings and captures of Atlantic sturgeon in the James River. However, this information has not been comprehensive enough to develop a population estimate for the James River or to provide sufficient evidence to confirm increased abundance. Some of the impact from the threats that facilitated the decline of the CB DPS have been removed (*e.g.*, directed fishing) or reduced as a result of improvements in water quality since passage of the Clean Water Act (CWA). As described in Section 4.4, we have estimated that there is a minimum ocean population of 8,811 CB DPS Atlantic sturgeon, of which 2,319 are adults and 6,608 are subadults of size vulnerable to capture in federal marine fisheries.

Areas with persistent, degraded water quality, habitat impacts from dredging, continued bycatch in U.S. state and federally managed fisheries, Canadian fisheries and vessel strikes remain significant threats to the CB DPS of Atlantic sturgeon. Of the 35% of Atlantic sturgeon incidentally caught in the Bay of Fundy, about 1% were CB DPS fish (Wirgin *et al.* 2012). Studies have shown that Atlantic sturgeon can only sustain low levels of bycatch mortality (Boreman 1997; ASMFC 2007; Kahnle *et al.* 2007). The CB DPS is currently at risk of extinction given (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and, (3) the impacts and threats that have and will continue to affect the potential for population recovery.

### 4.4.4 Status of the Carolina DPS

The Carolina DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor. The marine range of Atlantic sturgeon from the Carolina DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the Carolina DPS and the adjacent portion of the marine range are shown in Figure 1. Sturgeon are commonly captured 40 miles offshore (D. Fox, DSU, pers. comm.). Records providing fishery bycatch data by depth show the vast majority of Atlantic sturgeon bycatch via gillnets is observed in waters less than 50 meters deep (Stein *et al.* 2004b, ASMFC 2007), but Atlantic sturgeon are recorded as bycatch out to 500 fathoms.

Rivers known to have current spawning populations within the range of the Carolina DPS include the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Pee Dee Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed or mature adults were present in freshwater portions of a system (Table 16). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. There may also be spawning populations in the Neuse, Santee and Cooper Rivers, though it is uncertain. Historically, both the Sampit and Ashley Rivers were documented to have spawning populations at one time. However, the spawning population in the Sampit River is believed to be extirpated, and the current status of the spawning population in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. Fish from the Carolina DPS likely use other river systems than those listed here for their specific life functions.

**Table 16** Major rivers, tributaries, and sounds within the range of the Carolina DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

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

Historical landings data indicate that between 7,000 and 10,500 adult female Atlantic sturgeon were present in North Carolina prior to 1890 (Armstrong and Hightower 2002, Secor 2002). Secor (2002) estimates that 8,000 adult females were present in South Carolina during that same time-frame. Prior reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the Carolina DPS. Currently, the Atlantic sturgeon spawning population in at least one river system within the Carolina DPS has been extirpated, with potential extirpation in an additional system. The abundances of the remaining river populations within the DPS, each estimated to have fewer than 300 spawning adults, are estimated to be less than 3% of what they were historically (ASSRT 2007). As described in Section 4.4, we have estimated that there are a minimum of 1,356 Carolina DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries.

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

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

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations in the Southeast in the mid to late 19$^{th}$ century, from which they have never rebounded. Continued bycatch of Atlantic sturgeon in commercial fisheries is an ongoing impact to the Carolina DPS. More robust fishery independent data on bycatch are available for the northeast and mid-Atlantic than in the Southeast where high levels of bycatch underreporting are suspected.

Though there are statutory and regulatory regulations that authorize reducing the impact of dams on riverine and anadromous species, these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Water quality continues to be a problem in the Carolina DPS, even with existing controls on some pollution sources. Current regulatory regimes are not effective in controlling water allocation issues (e.g., no restrictions on interbasin water transfers in South Carolina, the lack of ability to regulate non-point source pollution, etc.).

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

The concept of a viable population able to adapt to changing environmental conditions is critical to Atlantic sturgeon, and the low population numbers of every river population in the Carolina DPS put them in danger of extinction throughout their range; none of the populations are large or stable enough to provide with any level of certainty for continued existence of Atlantic sturgeon in this part of its range. Although the largest impact that caused the precipitous decline of the species has been curtailed (directed fishing), the population sizes within the Carolina DPS have remained relatively constant at greatly reduced levels (approximately 3% of historical population sizes) for 100 years. Small numbers of individuals resulting from drastic reductions in populations, such as that which occurred due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry 1971; Shaffer 1981; Soulé 1980). Recovery of depleted populations is an inherently slow process for late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, it also increases the time frame over which exposure to the multitude of threats facing the Carolina DPS can occur. The viability of the Carolina DPS depends on having multiple self-sustaining riverine spawning populations and maintaining suitable habitat to support the various life functions (spawning, feeding, growth) of Atlantic sturgeon populations.

*Summary of the Status of the Carolina DPS of Atlantic Sturgeon*
Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon. Their late age at maturity provides more opportunities for individuals to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, this is hampered within the Carolina DPS by habitat alteration and bycatch. This DPS was severely depleted by past directed commercial fishing, and faces ongoing impacts and threats from habitat alteration or inaccessibility, bycatch, and the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch that have prevented river populations from rebounding and will impede their recovery.

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

### 4.4.5 Status of South Atlantic DPS

The South Atlantic DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) of the Ashepoo, Combahee, and

Edisto Rivers (ACE) Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, FL. The marine range of Atlantic sturgeon from the South Atlantic DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the South Atlantic DPS and the adjacent portion of the marine range are shown in Figure 1. Sturgeon are commonly captured 40 miles offshore (D. Fox, DSU, pers. comm.). Records providing fishery bycatch data by depth show the vast majority of Atlantic sturgeon bycatch via gillnets is observed in waters less than 50 meters deep (Stein *et al.* 2004b, ASMFC 2007), but Atlantic sturgeon are recorded as bycatch out to 500 fathoms (900 meters).

Rivers known to have current spawning populations within the range of the South Atlantic DPS include the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed, or mature adults were present, in freshwater portions of a system (Table 17). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. Historically, both the Broad-Coosawatchie and St. Marys Rivers were documented to have spawning populations at one time; there is also evidence that spawning may have occurred in the St. Johns River or one of its tributaries. However, the spawning population in the St. Marys River, as well as any historical spawning populations present in the St. Johns, are believed to be extirpated, and the status of the spawning population in the Broad-Coosawatchie is unknown. Both the St. Marys and St. Johns Rivers are used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. The use of the Broad-Coosawatchie by sturgeon from other spawning populations is unknown at this time. The presence of historical and current spawning populations in the Ashepoo River has not been documented; however, this river may currently be used for nursery habitat by young Atlantic sturgeon originating from other spawning populations. Fish from the South Atlantic DPS likely use other river systems than those listed here for their specific life functions.

**Table 17:** Major rivers, tributaries, and sounds within the range of the South Atlantic DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

| River/Estuary | Spawning Population | Data |
|---|---|---|
| ACE (Ashepoo, Combahee, and Edisto Rivers) Basin, SC; St. Helena Sound | Yes | 1,331 YOY (1994-2001); gravid female and running ripe male in the Edisto (1997); 39 spawning adults (1998) |
| Broad-Coosawhatchie Rivers, SC; Port Royal Sound | Unknown | |
| Savannah River, SC/GA | Yes | 22 YOY (1999-2006); running ripe male (1997) |
| Ogeechee River, GA | Yes | age-1 captures, but high inter-annual variability (1991-1998); 17 YOY (2003); 9 YOY (2004) |
| Altamaha River, GA | Yes | 74 captured/308 estimated spawning adults (2004); 139 captured/378 estimated spawning adults (2005) |
| Satilla River, GA | Yes | 4 YOY and spawning adults (1995-1996) |
| St. Marys River, GA/FL | Extirpated | |
| St. Johns River, FL | Extirpated | |

The riverine spawning habitat of the South Atlantic DPS occurs within the South Atlantic Coastal Plain ecoregion, which includes fall-line sandhills, rolling longleaf pine uplands, wet pine flatwoods, isolated depression wetlands, small streams, large river systems, and estuaries. Other ecological systems in the ecoregion include maritime forests on barrier islands, pitcher plant seepage bogs, and Altamaha grit (sandstone) outcrops. The primary threats to biological diversity in the South Atlantic Coastal Plain listed by The Nature Conservancy (TNC) are intensive silvicultural practices, including conversion of natural forests to highly managed pine monocultures and the clear-cutting of bottomland hardwood forests. Changes in water quality and quantity caused by hydrologic alterations (impoundments, groundwater withdrawal, and ditching), and point and nonpoint pollution, are threatening the aquatic systems. Development is a growing threat, especially in coastal areas. Agricultural conversion, fire regime alteration, and the introduction of nonnative species are additional threats to the ecoregion’s diversity. The South Atlantic DPS’s spawning rivers, located in the South Atlantic Coastal Plain, are primarily of two types: brownwater (with headwaters north of the Fall Line, silt-laden) and blackwater (with headwaters in the coastal plain, stained by tannic acids).

Secor (2002) estimates that 8,000 adult females were present in South Carolina before the collapse of the fishery in 1890. However, because fish from South Carolina are included in both the Carolina and South Atlantic DPSs, it is likely that some of the historical 8,000 fish would be attributed to both the Carolina DPS and the South Atlantic DPS. The sturgeon fishery had been the third largest fishery in Georgia. Reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the South Atlantic DPS. Currently, the Atlantic sturgeon population in at least two river systems within the South Atlantic DPS has been extirpated. As described in Section 4.4, we have estimated that there are a minimum of 14,911 SA DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries.

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

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

The directed Atlantic sturgeon fishery caused initial severe declines in southeast Atlantic sturgeon populations. Although the directed fishery is closed, bycatch in other commercial fisheries continues to impact the South Atlantic DPS. Statutory and regulatory mechanisms exist that authorize reducing the impact of dams on riverine and anadromous species such as Atlantic sturgeon, but these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Further, water quality continues to be a problem in the South Atlantic DPS, even with existing controls on some pollution sources. Current regulatory regimes are not effective in controlling water allocation issues (*e.g.,* no permit requirements for water withdrawals under 100,000 gpd in Georgia, no restrictions on interbasin water transfers in South Carolina, the lack of ability to regulate non-point source pollution.)

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

*Summary of the Status of the South Atlantic DPS of Atlantic Sturgeon*
The population of mature adult Atlantic sturgeon in the South Atlantic DPS is estimated to be at least 3,728. The DPS's freshwater range occurs in the watersheds (including all rivers and tributaries) of the ACE Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, FL. Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon. Their late age at maturity provides more opportunities for individuals to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, this is hampered within the South Atlantic DPS by habitat alteration, bycatch, and from the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch.

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

### 4.5 Status of Gulf of Maine DPS of Atlantic Salmon

Atlantic salmon are an anadromous species that typically spends two years at sea before returning to natals rivers to spawn. Juvenile salmon typically spend two years in fresh water before migrating to the ocean. Atlantic salmon are native to the North Atlantic Ocean. They range from the Connecticut River in the United States to Ungava Bay in northern Quebec, Canada in the western North Atlantic, and from Portugal to the Kola Penninsula in northwestern Russia in the eastern North Atlantic (Scott and Crossman 1973). In the United States, Atlantic salmon historically ranged from Maine south to Long Island Sound. However, the Central New England DPS and Long Island Sound DPS have both been extirpated (65 FR 69459; Nov. 17, 2000).

The GOM DPS of anadromous Atlantic salmon was initially listed by USFWS and NMFS (collectively, the Services) as an endangered species on November 17, 2000 (65 FR 69459). A subsequent review by the Services (74 FR 29344; June 19, 2009) included an expanded range for the GOM DPS of Atlantic salmon based on an interagency Status Review (Fay *et al.* 2006). Fay *et al.* (2006) concluded that all salmon populations inhabiting the large and small rivers from the Androscoggin River northward to the Dennys River differ genetically and in important life history characteristics from Atlantic salmon in adjacent portions of Canada (Spidle *et al.* 2003; Fay *et al.* 2006). Thus, Fay *et al.* (2006) concluded that this expanded group of populations (a "distinct population segment") met both the discreteness and significance criteria of NMFS and USFWS's DPS Policy (61 FR 4722; Feb. 7, 1996). The final rule agreed with the conclusions of BRT regarding the DPS delineation of Maine Atlantic salmon (74 FR 29344, June 19, 2009).

The GOM DPS includes all anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, and wherever these fish occur in the estuarine and marine environment. The marine range of the GOM DPS extends from the Gulf of Maine, throughout the Northwest Atlantic Ocean, to the coast of Greenland.

Included in the GOM DPS are all associated conservation hatchery populations used to supplement these natural populations; currently, such conservation hatchery populations are maintained at Green Lake National Fish Hatchery (GLNFH) and Craig Brook National Fish Hatcheries (CBNFH), both operated by the USFWS, as well as private watershed-based facilities (Downeast Salmon Federation's East Machias and Pleasant River facilities). Excluded from the GOM DPS are landlocked Atlantic salmon and those salmon raised in commercial hatcheries for the aquaculture industry (74 FR 29344; June 19, 2009).

### 4.5.1 Species Description

Atlantic salmon have a complex life history that includes territorial rearing in rivers to extensive feeding migrations on the high seas. During their life cycle, Atlantic salmon go through several distinct phases that are identified by specific changes in behavior, physiology, morphology, and habitat requirements.

Adult Atlantic salmon return to rivers from the ocean and migrate to their natal streams to spawn. Adults ascend the rivers within the GOM DPS beginning in the spring. The ascent of adult salmon continues into the fall. Although spawning does not occur until late fall, the majority of Atlantic salmon in Maine enter freshwater between May and mid-July (Meister 1958; Baum 1997). Early migration is an adaptive trait that ensures adults have sufficient time to effectively reach spawning areas despite the occurrence of temporarily unfavorable conditions that naturally occur within rivers (Bjornn and Reiser 1991). Salmon that return in early spring spend nearly five months in the river before spawning, often seeking cool water refuge (e.g., deep pools, springs, and mouths of smaller tributaries) during the summer months.

In the fall, female Atlantic salmon select sites for spawning. Spawning sites are positioned within flowing water, particularly where upwelling of groundwater occurs, allowing for percolation of water through the gravel (Danie *et al.*1984). These sites are most often positioned at the head of a riffle (Beland *et al.* 1982); the tail of a pool; or the upstream edge of a gravel bar where water depth is decreasing and water velocity is increasing (McLaughlin and Knight 1987; White 1942), and hydraulic head allows for permeation of water through the redd (a gravel depression where eggs are deposited). Females produce a total of 1,500 to 1,800 eggs per kilogram of body weight, yielding an average of 7,500 eggs per two sea-winter (SW) female (an adult female that has spent two winters at sea before returning to spawn) (Baum and Meister 1971). After spawning, Atlantic salmon may either return to sea immediately or remain in fresh water until the following spring before returning to the sea (Fay *et al.* 2006).

Embryos develop in the redd for a period of 175 to 195 days, hatching in late March or April (Danie *et al.*1984). Newly hatched salmon referred to as larval fry, alevin, or sac fry, remain in the redd for approximately six weeks after hatching and are nourished by their yolk sac (Gustafson-Greenwood and Moring 1991). Survival from the egg to fry stage in Maine is estimated to range from 15 to 35% (Jordan and Beland 1981). When fry reach approximately 4 cm in length, the young salmon are termed parr (Danie *et al.* 1984). Most parr remain in the river for two to three years before undergoing smoltification, the process in which parr go through physiological changes in order to transition from a freshwater environment to a saltwater marine environment. Some male parr may not go through smoltification and will become sexually mature and participate in spawning with sea-run adult females. These males are referred to as "precocious parr." During the smoltification process, parr markings fade and the body becomes streamlined and silvery with a pronounced fork in the tail. Naturally reared smolts in Maine range in size from 13 to 17 cm, and most smolts enter the sea during May to begin their first ocean migration (USASAC 2004).

The spring migration of smolts to the marine environment takes 25 to 45 days for an entire population to emigrate. Individual smolts move relatively rapidly, exiting the estuary within several tidal cycles (Hyvarinen *et al.* 2006; Lacroix and McCurdy 1996; Lacroix *et al.* 2004, 2005). Smolts are termed postsmolts after from ocean entry to the end of the first winter at sea (Allan and Ritter 1977). Post-smolts generally travel out of coastal systems on the ebb tide and may be delayed by flood tides (Hyvarinen *et al.* 2006; Lacroix and McCurdy 1996; Lacroix *et al.* 2004, 2005). Lacroix and McCurdy (1996), however, found that postsmolts exhibit active, directed swimming in areas with strong tidal currents. Studies in the Bay of Fundy and Passamaquoddy Bay suggest some aggregation and common migration corridors related to surface currents (Hyvarinen *et al.* 2006; Lacroix and McCurdy 1996; Lacroix *et al.* 2004). Postsmolt distribution may reflect water temperatures (Reddin and Shearer 1987) and/or the major surface-current vectors (Lacroix and Knox 2005). Postsmolts travel mainly at the surface of the water column (Renkawitz *et al.* 2012) and may form shoals, possibly of fish from the same river (Shelton *et al.* 1997). Post-smolts grow quickly, achieving lengths of 12-14 inches by October (Baum 1997).

During the late summer and autumn of the first year, North American post-smolts are concentrated in the Labrador Sea and off of the west coast of Greenland, with the highest concentrations between 56$^{\circ}$ N. and 58$^{\circ}$N. (Reddin 1985; Reddin and Short 1991; Reddin and Friedland 1993, Sheehan *et al.* 2012). The salmon located off Greenland are primarily composed of non-maturing 1SW fish destined to spawn as 2SW fish from both North America and Europe, plus a smaller component of previous spawners who have returned to the sea prior to their next spawning event (Reddin 1988; Reddin *et al.* 1988). The following spring, 1SW and older fish are generally located in the Gulf of St. Lawrence, off the coast of Newfoundland, and on the east coast of the Grand Banks (Reddin 1985; Dutil and Coutu 1988; Ritter 1989; Reddin and Friedland 1993; and Friedland *et al.* 1999).

Some salmon may remain at sea for another year or more before maturing. After their second winter at sea, the salmon likely over-winter in the area of the Grand Banks before returning to their natal rivers to spawn (Reddin and Shearer 1987). Reddin and Friedland (1993) found non-maturing adults located along the coasts of Newfoundland, Labrador, and Greenland, and in the Labrador and Irminger Sea in the later summer and autumn.

#### 4.5.2 Disease and Predators

Many parasites and diseases are known to infect Atlantic salmon, but Maine wild salmon populations have not had documented outbreaks (Baum 1997). Most of the infections occur under hatchery or other crowded rearing conditions. The common sea louse, found only on salmonids, is prevalent on Atlantic salmon at sea. The common brook trout ecto-parasite has been occasionally observed on juvenile salmon in Maine rivers. In salt water, vibriosis is a common bacterial disease affecting most species of fish, including farmed Atlantic salmon. Vibriosis is also thought to affect wild salmon populations (Baum 1997).

The retrovirus salmon swimbladder sarcoma virus (SSSV) appears to exist at some level in wild populations of salmon in Maine, although symptoms have not been observed in wild salmon (AASBRT 1999). In 1998, SSSV was detected in Pleasant River broodstock held by the USFWS, resulting in the decision to destroy all captive broodstock for this river. SSSV has been identified at very low levels in captive broodstock populations from three other GOM DPS rivers.

Coldwater disease is caused by the bacterium *Flavobacterium psychrophilum* and has recently been found to be a serious problem for Atlantic salmon in New England waters. The pathogen causes mortality in juvenile salmon. The pathogen is transmitted vertically from carrier sea-run adults to offspring via eggs [U.S. Atlantic Salmon Assessment Committee (USASAC) 2000; 65 FR 69476, Nov. 17, 2000)].

The infectious salmon anemia virus (ISAV) appeared on the North American continent in 1996 in Canadian aquaculture pens, within the known infective range of U.S. sea pens. ISAV was first detected at a Maine salmon farm in Cobscook Bay in January 2001, with subsequent outbreaks at several other salmon farms in Cobscook Bay. On December 18, 2001, the U.S. Department of Agriculture's (USDA) Animal and Plant Health Inspection Service (APHIS) implemented an ISAV indemnity, surveillance, biosecurity, and epidemiological research program for farm-raised fish in the United States. Participation in this program is mandatory for all salmon growers and covers all salmon finfish farms in the state. USDA's goal is to control and contain the disease through rapid detection and depopulation of salmon that have been infected with or exposed to ISAV.

On January 7, 2002, the Maine DMR and the APHIS ordered the eradication of up to 1.5 million salmon located in seven aquaculture facilities in Cobscook Bay that were infected with or exposed to the ISAV. The January 2002 order followed the earlier removal of over one million ISAV-exposed fish by the aquaculture industry, as directed by the MDMR. The fish were removed from Cobscook Bay and the entire bay was fallowed for ninety-two days. Amplification of endemic diseases, such as ISAV, poses a threat to wild populations of salmon, but continued surveillance and monitoring programs should reduce the risk of future outbreaks within the aquaculture industry and therefore reduce the risk of transmission of ISAV to wild salmon. The ISAV virus is extremely destructive to maturing salmon, and there is no known cure (USASAC 2000; 65 FR 69476, Nov. 17, 2000). Detection of the pathogen has continued in the United States and Canada since the initial outbreak, but good fish husbandry and diligent surveillance and monitoring has kept the disease from emerging in U.S. commercial salmon farms since 2006 (see below). However, recent reports (2012) of non-pathogenic strains of ISAV have occurred on some farm sites in Canada. Furthermore, the U.S. is working with Canada on joint strategies for managing ISAV, recognizing the importance of working together on issues affecting a common water body. The APHIS program is being interfaced with the State of Maine's husbandry and bay management program that is being implemented via the Maine DMR's authority described in the previous sections above. Additional surveillance by the APHIS and the MDMR includes tracking of the following: the dispersion of the virus in the water column; the attenuation of the virus on surfaces over time; and the environmental distribution of the virus in the water column, sediments, alternative species, and sea lice. These programs developed by the USDA APHIS and the MDMR to address outbreaks of ISAV in the aquaculture industry should reduce the threat of this disease to wild salmon.

Known predators of Atlantic salmon include marine mammals (e.g., seals, porpoises, and dolphins), terrestrial mammals (e.g., otters, minks), birds, fish and sharks. Atlantic salmon post-smolts are preyed upon by cod, whiting, cormorants, ducks, terns, gulls, and many other opportunistic predators (Hvidsten and Møkkelgjerd 1987; Gunnerød *et al.* 1988; Hvidsten and Lund 1988; Montevecchi *et al.* 1988; Hislop and Shelton 1993). Cormorants and striped bass are transitory predators that impact migrant juveniles in the lower river, estuarine, and coastal areas areas. Seals have reached high population levels, and salmon remain vulnerable to seal predation throughout much of their marine migration range.

#### 4.5.3 Status and Trends of Atlantic Salmon

The abundance of Atlantic salmon within the range of the GOM DPS has been generally declining since the 1800s (Fay *et al.* 2006). A comprehensive time series of adult returns to the GOM DPS dating back to 1967 exists (Fay *et al.* 2006, USASAC 2013) (Figure 3). It is important to note that contemporary abundance levels of Atlantic salmon within the GOM DPS are several orders of magnitude lower than historical abundance estimates. For example, Foster and Atkins (1869) estimated that roughly 100,000 adult salmon returned to the Penobscot River alone before the river was dammed, whereas contemporary estimates of abundance for the entire GOM DPS have rarely exceeded 5,000 individuals in any given year since 1967 (Fay *et al.* 2006, USASAC 2013).

Contemporary abundance estimates are informative in considering the conservation status of the GOM DPS today. After a period of population growth between the 1970s and the early 1980s, adult returns of salmon in the GOM DPS peaked between approximately 1984 and 2001 before declining during the 2000s. Adult returns fluctuated over the last few years, with increases observed from 2008 to 2011, and a decrease again in 2012. The population growth observed in the 1970s is likely attributable to favorable marine survival and increases in hatchery capacity, particularly from GLNFH that was constructed in 1974. Marine survival remained relatively high throughout the 1980s, and salmon populations in the GOM DPS remained relatively stable until the early 1990s. In the early 1990s, marine survival rates decreased, leading to the declining trend in adult abundance observed throughout 1990s and early 2000s. The increase in the abundance of returning adult salmon observed between 2008 and 2011 may be an indication of improving marine survival; however the decline in 2012 may suggest otherwise.

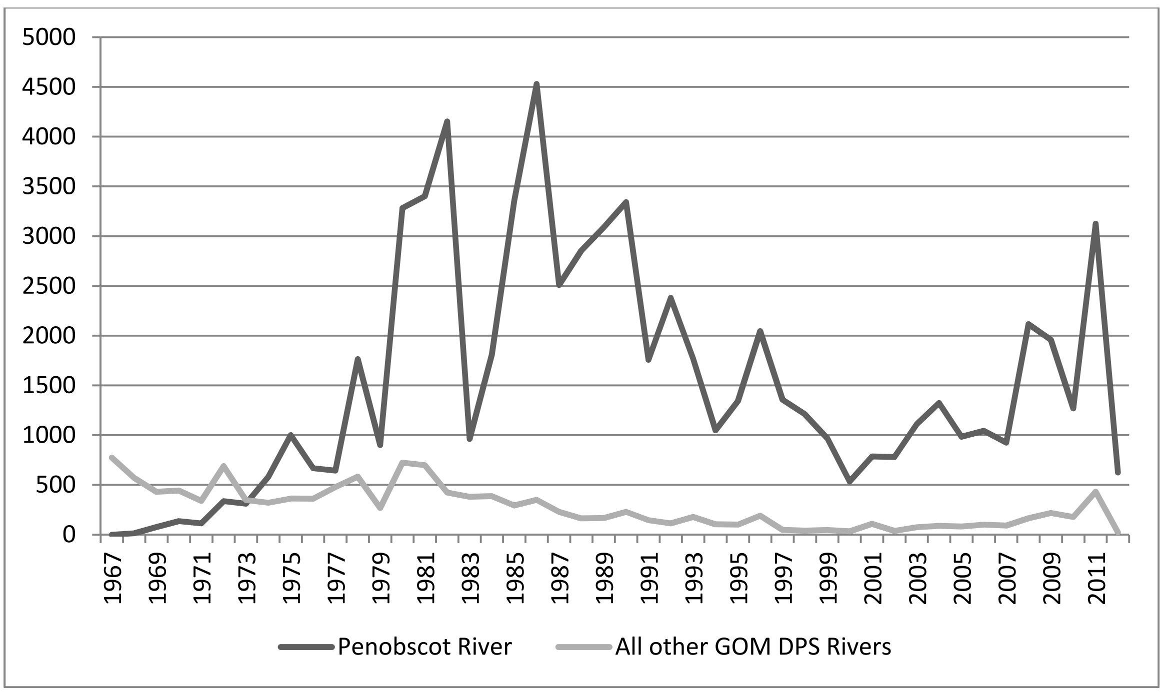


**Figure 3** Adult returns to the GOM DPS Rivers between 1967 and 2012 (Fay *et al.* 2006, USASAC 2013).

Adult returns to the GOM DPS have been very low for many years and remain extremely low in terms of adult abundance in the wild. Further, the majority of all adults in the GOM DPS return to a single river, the Penobscot, which accounted for 91% of all adult returns to the GOM DPS between 2000 and 2011. Of the 3,125 adult returns to the Penobscot in 2011, the vast majority are the result of smolt stocking; and only a small portion were naturally reared. The term naturally reared includes fish originating from both natural spawning and from stocked hatchery fry (USASAC 2012). Hatchery fry are included as naturally reared because hatchery fry are not marked, and therefore cannot be distinguished from fish produced through natural spawning. Low abundances of both hatchery-origin and naturally reared adult salmon returns to Maine demonstrate continued poor marine survival.

The abundance of Atlantic salmon in the GOM DPS has been low and either stable or declining over the past several decades. The proportion of fish that are of natural origin is very small (approximately 6% over the last ten years) but appears stable. The conservation hatchery program has assisted in slowing the decline and helping to stabilize populations at low levels. However, stocking of hatchery products has not contributed to an increase in the overall abundance of salmon and as yet has not been able to increase the naturally reared component of the GOM DPS. Continued reliance on the conservation hatchery program could prevent extinction in the short term, but recovery of the GOM DPS must be accomplished through increases in naturally reared salmon.

#### 4.5.4 Factors Affecting Recovery of Atlantic Salmon

There are a wide variety of factors that have and continue to affect the current status of the GOM DPS. The potential interactions among these factors are not well understood, nor are the reasons for the limited response of salmon populations to the many ongoing conservation efforts for this species.

The recovery plan for the previously designated GOM DPS (NMFS and USFWS 2005), the latest status review (Fay *et al.* 2006), and the 2009 listing rule all provide a comprehensive assessment of the many factors, including both threats and conservation actions, that are currently affecting the status and recovery of listed Atlantic salmon. The Services are updating the recovery plan to include the current GOM DPS and its designated critical habitat. The new recovery plan provides the most up to date list of significant threats affecting the GOM DPS. These are the following:

- Dams
- Inadequacy of existing regulatory mechanisms for dams
- Continued low marine survival rates for U.S. stocks of Atlantic salmon
- Lack of access to spawning and rearing habitat due to dams and road-stream crossings

In addition to these significant threats, there are a number of lesser stressors identified in the recovery plan. These are the following:

- Degraded water quality
- Aquaculture practices, which pose ecological and genetic risks
- Climate change
- Depleted diadromous fish communities
- Incidental capture of adults and parr by recreational anglers
- Introduced fish species that compete or prey on Atlantic salmon
- Poaching of adults
- Recovery hatchery program (potential for artificial selection/domestication)

- Sedimentation of spawning and rearing habitat
- Water extraction

Findings in Fay *et al.* (2006) supported the determination that each of the five listing factors is at least partly responsible for the present low abundance of the GOM DPS. This is reflected in and supplemented by the final listing rule for the new GOM DPS (NMFS and USFWS 2009). The following gives a brief overview of the five ESA factors as related to the GOM DPS.

1. **Present or threatened destruction, modification, or curtailment of its habitat or range –** Historically and, to a lesser extent currently, dams have adversely impacted Atlantic salmon by obstructing fish passage and degrading riverine habitat. Dams are considered to be one of the primary causes of both historic declines and the contemporary low abundance of the GOM DPS. Land use practices, including forestry and agriculture, have reduced habitat complexity (e.g., removal of large woody debris from rivers) and habitat connectivity (e.g., poorly designed road crossings) for Atlantic salmon. Water withdrawals, elevated sediment levels, and acid rain also degrade Atlantic salmon habitat.

2. **Overutilization for commercial, recreational, scientific, or educational purposes –** While most directed commercial fisheries for Atlantic salmon have ceased, the impacts from past fisheries are still important in explaining the present low abundance of the GOM DPS. Both poaching and bycatch in recreational and commercial fisheries for other species remain of concern, given critically low numbers of salmon.

3. **Predation and disease –** Natural predator-prey relationships in aquatic ecosystems in the GOM DPS have been substantially altered by introduction of non-native fishes (e.g., chain pickerel, smallmouth bass, and northern pike), declines of other native diadromous fishes, and alteration of habitat by impounding free-flowing rivers and removing instream structure (such as removal of boulders and woody debris during the log-driving era). The threat of predation on the GOM DPS is noteworthy because of the imbalance between the very low numbers of returning adults and the recent increase in populations of some native predators (e.g., double-crested cormorant), as well as non-native predators. Atlantic salmon are susceptible to a number of diseases and parasites, but mortality is primarily documented at conservation hatcheries and aquaculture facilities.

4. **Inadequacy of existing regulatory mechanisms –** The ineffectiveness of current federal and state regulations at requiring fish passage and minimizing or mitigating the aquatic habitat impacts of dams is a significant threat to the GOM DPS today. Furthermore, most dams in the GOM DPS do not require state or federal permits. Although the State of Maine has made substantial progress in regulating water withdrawals for agricultural use, threats still

remain within the GOM DPS, including those from the effects of irrigation wells on salmon streams.

5. **Other natural or manmade factors –** Poor marine survival rates of Atlantic salmon are a significant threat, although the causes of these decreases are unknown. The role of ecosystem function among the freshwater, estuarine, and marine components of the Atlantic salmon's life history, including the relationship of other diadromous fish species in Maine (e.g., American shad, alewife, sea lamprey), is receiving increased scrutiny in its contribution to the current status of the GOM DPS and its role in recovery of the Atlantic salmon. While current state and federal regulations pertaining to finfish aquaculture have reduced the risks to the GOM DPS (including eliminating the use of non-North American Atlantic salmon and improving containment protocols), risks from the spread of diseases or parasites and from farmed salmon escapees interbreeding with wild salmon still exist.

Efforts aimed at protecting Atlantic salmon and their habitats in Maine have been underway for well over 100 years. These efforts are supported by a number of federal, state, and local government agencies, as well as many private conservation organizations. The 2005 recovery plan for the originallylisted GOM DPS (NMFS and USFWS 2005) presented a strategy for recovering Atlantic salmon that focused on reducing the most severe threats to the species and immediately halting the decline of the species to prevent extinction. The 2005 recovery program included the following elements:

1. Protect and restore freshwater and estuarine habitats;
2. Minimize potential for take in freshwater, estuarine, and marine fisheries;
3. Reduce predation and competition for all life-stages of Atlantic salmon;
4. Reduce risks from commercial aquaculture operations;
5. Supplement wild populations with hatchery-reared DPS salmon;
6. Conserve the genetic integrity of the DPS;
7. Assess stock status of key life stages;
8. Promote salmon recovery through increased public and government awareness; and
9. Assess effectiveness of recovery actions and revise as appropriate.

A wide variety of activities have focused on protecting Atlantic salmon and restoring the GOM DPS, including (but not limited to) hatchery supplementation; removing dams or providing fish passage; improving road crossings that block passage or degrade stream habitat; protecting riparian corridors along rivers; reducing the impact of irrigation water withdrawals; limiting effects of recreational and commercial fishing; reducing the effects of finfish aquaculture; outreach and education activities; and research focused on better understanding the threats to Atlantic salmon and developing effective restoration strategies. In light of the 2009 GOM DPS listing and designation of critical habitat, the Services are producing a new recovery plan for the expanded GOM DPS of Atlantic salmon.

The final rule designating critical habitat for the GOM DPS identifies a number of activities that have and will likely continue to impact the biological and physical features of spawning, rearing, and migration habitat for Atlantic salmon. These include agriculture, forestry, changing land-use and development, hatcheries and stocking, roads and road-crossings and other instream activities (such as alternative energy development), mining, dams, dredging, and aquaculture. Most of these activities have or still do occur, at least to some extent, in the three Salmon Habitat Recovery Units (SHRU): the Merrymeeting Bay SHRU, the Penobscot Bay SHRU, and the Downeast SHRU.

Today, dams are the greatest impediment, outside of marine survival, to the recovery of salmon in the Penobscot, Kennebec, and Androscoggin river basins (Fay *et al.* 2006). Hydropower dams in the Merrymeeting Bay SHRU significantly impede the migration of Atlantic salmon and other diadromous fish and either reduce or eliminate access to roughly 352,000 units of historically accessible spawning and rearing habitat. In addition to hydropower dams, agriculture and urban development largely affect the lower third of the Merrymeeting Bay SHRU by reducing substrate and cover, reducing water quality, and elevating water temperatures. Additionally, smallmouth bass and brown trout introductions, along with other non-indigenous species, significantly degrade habitat quality throughout the Merrymeeting Bay SHRU by altering natural predator/prey relationships.

Impacts to substrate and cover, water quality, water temperature, biological communities, and migratory corridors, among a host of other factors, have impacted the quality and quantity of habitat available to Atlantic salmon populations within the Downeast Coastal SHRU. Two hydropower dams on the Union river, and, to a lesser extent, the small ice dam on the lower Narraguagus River, limit access to roughly 18,500 units of spawning and rearing habitat within these two watersheds. In the Union River, which contains over 12,000 units of spawning and rearing habitat, physical and biological features have been most notably limited by high water temperatures and abundant smallmouth bass populations associated with impoundments. In the Pleasant River and Tunk Stream, which collectively contain over 4,300 units of spawning and rearing habitat, pH has been identified as possibly being the predominate limiting factor. The Machias, Narraguagus, and East Machias rivers contain the highest quality habitat relative to other HUC 10s in the Downeast Coastal SHRU and collectively account for approximately 40 percent of the spawning and rearing habitat in the Downeast Coastal SHRU.

*Summary of the Status of GOM DPS of Atlantic Salmon*

The last commercial fishery for Atlantic salmon in the U.S. was closed in 1948 (Fay *et al.* 2006). The GOM DPS of Atlantic salmon is endangered and includes anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River. The protections of the ESA apply wherever these fish occur, whether in rivers, estuaries, or the marine environment. Hatchery fish used to supplement these natural populations are also included under this listing.

Abundance levels of the GOM DPS of Atlantic salmon are at very low levels. Documented adult returns to the GOM DPS in 2012 were 939 (USASAC 2013). The proportion of fish that are of natural origin is very small (approximately 10%). The conservation hatchery program has assisted in slowing the decline and helping to stabilize populations at low levels, but has not contributed to an increase in the overall abundance of salmon. Regulations have been in place since 1987 restricting ocean harvest, and possession of Atlantic salmon is currently prohibited in EEZ waters under the federal FMP for Atlantic salmon (NEFMC 1987). Possession is also prohibited in riverine and coastal waters through complementary management measures enacted at the state level.

# 5.0 Environmental Baseline

Environmental baselines for biological opinions include the past and present impacts of all state, federal, or private actions, as well as any other human activities in the action area, the anticipated impacts of all proposed federal projects, including fisheries, in the action area that have already undergone formal or early section 7 consultation, and the impacts of state or private actions that are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this Opinion includes the effects of several activities that may affect the survival and recovery of right, humpback, fin and sei whales, loggerhead, leatherback, Kemp’s ridley, and green sea turtles, and ESA-listed fish in the action area.

## 5.1 Federal Actions with Formal or Early Section 7 Consultations

ESA section 7 consultation has been conducted on all federal fisheries authorized under a federal fishery management plan, as well as on other federal actions (i.e., dredging, research activities, vessel activities, etc.).

The effects of federal fisheries on the prey and habitat of ESA-listed cetaceans, sea turtles, and fish are expected to be discountable, as are the effects of vessels involved in fishing activities, as discussed briefly below. Sections 5.1.1 and 5.1.2 then discuss the effects that federal fishing activities have had and continue to have on ESA-listed species, while sections 5.1.3, 5.1.4, and 5.1.5 discuss the baseline effects of other federal actions.

As described in the Status of the Species sections (4.2.1-4.2.4), large whales consume copepods, krill, and/or small schooling fish. Copepods and krill are generally too small to become entrapped in fishing gear or affected by commercial fishing activity. Schooling fish, such as herring and mackerel, are targeted by fishermen, but given the diversity of humpback and fin whale diets, commercial fishery operations are not expected to have a significant effect on the availability of whale prey species.

Some sea turtle prey items—horseshoe crabs, other crabs, whelks, and fish—are removed from the marine environment as fisheries directed catch and bycatch. None of these are typical prey species of leatherback sea turtles or of neritic juvenile or adult green sea turtles that inhabit continental shelf waters (Rebel 1974; Mortimer 1982; Bjorndal 1985; NMFS and USFWS 1992b; Bjorndal 1997). Neritic juveniles and adults of both loggerhead and Kemp's ridley sea turtles feed on these species (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993; Burke *et al.* 1994; Morreale and Standora 2005; Seney and Musick 2005). However, some of the bycatch is expected to be returned to the water alive, while the remainder will be returned to the water dead or mortally injured. Injured or deceased bycatch would still be available as prey for sea turtles, particularly loggerheads, which are known to eat a variety of live prey as well as scavenge dead organisms (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993; Morreale and Standora 2005). Sea turtles are not thought to be food-limited.

Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007). There is no indication that Atlantic sturgeon are food-limited or that commercial fisheries might negatively impact their food availability, given the diversity of their diets.

Diets of Atlantic salmon post-smolts include invertebrates, amphipods, euphausiids, and fish (Hislop and Youngson 1984; Jutila and Toivonen 1985; Fraser 1987; Hislop and Shelton 1993). As adults, Atlantic salmon primarily eat fish, feeding upon capelin, herring, and sand lance (Hansen and Pethon 1985; Reddin 1985; Hislop and Shelton 1993). There is no indication that Atlantic salmon are food-limited or that commercial fisheries might negatively impact their food availability, given the diversity of their diets.

Bottom habitat in the action area may be adversely affected by gear used in the fisheries (NMFS 2003a). A panel of experts has previously concluded that the effects of even lightweight otter trawl gear would include: (1) scraping or plowing of the doors on the bottom, sometimes creating furrows along their path; (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom; (3) removal or damage to benthic or demersal species; and (4) removal or damage to structure forming biota. The panel also concluded that the greatest impacts from otter trawls occur in high and low energy gravel habitats and in hard clay outcroppings, and that sand habitats were the least likely to be impacted (NREFHSC 2002). The action area does not include hard clay outcroppings, although gravel habitats may occur.

The foraging distribution of Kemp's ridley, loggerhead, and green sea turtles in Mid-Atlantic and New England waters, 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. Alterations of bottom habitat should not affect foraging right, humpback, fin and sei whales (Baumgartner *et al.* 2003; IWC 1992; Pace and Merrick 2008; Perry *et al.* 1999), but they may be temporarily disturbed by the use of bottom fishing gear.

Alterations of bottom habitat in estuaries and coastal areas could affect foraging Atlantic sturgeon, but the extent of any negative impacts is unknown. Fishing effort does not occur everywhere that Atlantic sturgeon forage, and there is no indication at this time that Atlantic sturgeon are food-limited. Atlantic sturgeon are known to aggregate in areas that overlap with fishing activity, but it is not clear that these aggregations are related to foraging. Because Atlantic sturgeon spawn in rivers, the gear deployed by the seven fisheries under discussion are not expected to have any effect on sturgeon spawning activity or on early life stages (e.g., young-of-the-year or juveniles that have not yet left the rivers).

Atlantic salmon in the ocean are pelagic and highly surface oriented (Kocik and Sheehan 2006, Renkawitz *et al.* 2012). The preferred habitat of post-smolt salmon in the open ocean is principally the upper 10 meters of the water column (Baum 1997, ICES SGBYSAL 2005), although there is evidence of forays into deeper water for shorter periods. Adult Atlantic salmon demonstrate a wider depth profile (ICES SGBYSAL 2005), but overall salmon tend to be distributed in the surface layer. Gear deployed by the seven fisheries under consideration may disrupt surface waters temporarily, but is not expected to have a lasting effect on Atlantic salmon marine habitat.

For these reasons and the lack of any evidence that fishing practices affect habitats in degrees that harm or harass ESA-listed species, we find that while continued fishing efforts may potentially alter benthic habitats, these alterations will be insignificant to ESA-listed species.

ESA-listed cetaceans and sea turtles are known to be killed and injured as a result of being struck by vessels, but, because fishing vessels operate at slow speeds, any effects to these species by fishing vessels is discountable. Vessel strikes of Atlantic sturgeon have been observed in the James River, with 11 reported strikes between 2005 and 2007 (ASSRT 2007). Because we do not know the percent of total vessel strikes that the observed mortalities represent or whether these vessel strikes occur at similar rates in other rivers, we are not able to quantify the number of Atlantic sturgeon likely killed as a result of vessel strikes. Vessel strikes of Atlantic salmon have not been reported as a threat to the species.

### 5.1.1 Federal Fisheries Not Part of the Proposed Action

ESA section 7 consultation has been conducted on all Northeast federal fisheries authorized under a federal fishery management plan. The American lobster, Atlantic herring, Atlantic mackerel/squid/Atlantic butterfish, Atlantic sea scallop, highly migratory species, red crab, tilefish fisheries, and southeast U.S. shrimp fisheries are known to operate in the action area and are likely to contribute some portion of the fishing effort that may adversely affect threatened and endangered species. The past and present effects of these fisheries are discussed below.

##### *5.1.1.1 American Lobster*

American lobster occurs within U.S. waters from Maine to Virginia. They are most abundant from Maine to New Jersey with abundance declining from north to south (ASMFC 1999). The ASMFC's Interstate Fishery Management Plan (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 Southern New England (SNE) stock area. In response, the ASMFC adopted Addendum 17 to its Interstate Fishery Management Plan for American Lobster in February 2012. This addendum serves as the first phase to rebuild the SNE stock by adopting measures intended to reduce fishing exploitation by 10 % beginning in 2013. The management measures include a requirement for lobstermen to v-notch all legal-sized egg-bearing lobsters in LCMAs 2, 4 and 5; a minimum size increase for lobster harvested in offshore LCMA 3; and various closed seasons in LCMAs 2, 4, 5 and 6. The ASMFC adopted Addendum 18 in August 2012, which contains measures to address latent (unfished) effort and reduce the overall number of traps allocated in LCMAs 2 and 3 to scale the fishery to the size of the SNE resource. Some management tools include trap reductions, trap banking, and controlled growth using plans specialized for each affected management area. The ASMFC expects that additional action through subsequent addenda will be needed to complete the SNE rebuilding plan. NMFS is involved in the development of Addendum 18 through participation on the ASMFC's Lobster Management Board and will address the ASMFC's recommendations for Federal action in Addendum 17. The trap reduction measures associated with these actions may benefit large whales and sea turtles by reducing the amount of gear (specifically buoy lines) in the water where whales and sea turtles also occur.

The American lobster fishery has been identified as causing injuries to and mortality of loggerhead and leatherback sea turtles as a result of entanglement in buoy lines of the pot/trap gear (NMFS 2012b). Loggerhead or leatherback sea turtles caught/wrapped in the buoy lines of lobster pot/trap gear can die as a result of forced submergence or incur injuries leading to death as a result of severe constriction of a flipper from the entanglement. Between 2002 and 2011, the lobster trap fishery in state waters caused at least 51 leatherback entanglements in the Northeast Region. All entanglements involved the vertical line of the gear. These verified/confirmed entanglements occurred in Maine, Massachusetts, Rhode Island, Connecticut, and New Jersey state waters from May through October (Northeast Region STDN database).

Given the seasonal distribution of loggerhead sea turtles in Mid-Atlantic and New England waters and the operation of the lobster fishery, loggerhead sea turtles are expected to overlap with the placement of lobster pot/trap gear in the fishery during the months of May through October in waters off of New Jersey through Massachusetts. Compared to loggerheads, leatherback sea turtles have a similar seasonal distribution in Mid-Atlantic and New England waters, but with a more extensive distribution in the Gulf of Maine (Shoop and Kenney 1992; James *et al.* 2005a). Therefore, leatherback sea turtles are expected to overlap with the placement of lobster pot/trap gear in the fishery during the months of May through October in waters off of New Jersey through Maine.

Given the distribution of lobster fishing effort, leatherback sea turtles are the most likely sea turtle to be affected since this species occurs regularly in Gulf of Maine waters. The most recent Opinion for this fishery, completed on August 3, 2012, 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 2012 Opinion (See **Error! Reference source not found.** below) (NMFS 2012b).

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; Henry *et al.* 2011; Waring *et al.* 2011; 73 FR 73032, December 1, 2008). Large whales are known to become entangled in lines associated with multiple gear types. For pot/trap gear, vertical lines attach buoys to the gear while groundline attach the pots/traps in series. Lines wrapped tightly around an animal can cut into the flesh that can lead to injuries, infection and death (Moore *et al.* 2004).

A right whale entanglement in pot/trap gear used in the inshore lobster fishery resulting in death occurred in 2001 (Waring *et al.* 2007). A mortality of a humpback whale in pot/trap gear in the state lobster fishery occurred in 2002 (Waring *et al.* 2007). Other mortalities and serious injuries to ESA-listed cetaceans as a result of pot/trap gear consistent of that used in the lobster fishery have occurred as reported in Moore *et al.* (2004), Johnson *et al.* (2005), and Henry *et al.* (2011). 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.

#### *5.1.1.2 Atlantic Herring*

Purse seines, midwater trawls (single), and pair trawls are the three primary gears involved in the Atlantic herring fishery (NEFMC 2006). The gear type accounting for the majority of herring landings changed over the ten-year period from 1995 to 2005 (NEFMC 2006). During the 1990s, purse seine and mid-water trawl gear accounted for the majority of annual herring landings. Since 2000, pair trawl gear has accounted for the majority of herring landings (NEFMC 2006). Warden (2011) reported an estimate of zero loggerheads in bottom otter trawl gear targeting herring.

An FMP for the Atlantic herring fishery was implemented on December 11, 2000. Three management areas, which may have different management measures, were established under the Herring FMP. Changes to the management of the herring fishery were made in 2007 with the implementation of Amendment 1 to the Herring FMP (72 FR 11252, March 12, 2007). These included making the herring fishery a limited access fishery (NEFMC 2006). As a result of these changes, effort in the fishery is expected to be reduced or constrained. The ASMFC's Atlantic Herring ISFMP provides measures for the management of the herring fishery in state waters that are complementary to the federal FMP. The most recent reinitiated (due to the Atlantic salmon listing) consultation on the herring fishery was completed on February 9, 2010. After review and evaluation of observer data (no observed takes of ESA-listed species, despite increased observer coverage in recent years) and information on where and when the fishery operates, we concluded the consultation informally due to the discountable nature of whale, sea turtle, or Atlantic salmon interactions.

#### *5.1.1.3 Atlantic Pelagic Fisheries for Swordfish, Tuna, Sharks, and Billfish (Highly Migratory Species)*

Atlantic pelagic fisheries for swordfish, tuna, sharks, and billfish (highly migratory species or HMS) 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. We reinitiated consultation on the pelagic longline component of this fishery as a result of exceeded incidental take levels for loggerhead and leatherback sea turtles (NMFS 2004a). The resulting Opinion stated the long-term continued operation of the pelagic longline fishery for tuna and swordfish was likely to jeopardize the continued existence of leatherback sea turtles, but RPAs were implemented allowing for the continued authorization of the fishery that would not jeopardize leatherbacks. In 2006, the Atlantic HMS pelagic longline fishery had an estimated 771.6 interactions with loggerhead sea turtles and 381.3 interactions with leatherback sea turtles (Garrison *et al.* 2009).

The most recent formal section 7 consultation on the continued authorization of the Atlantic shark fisheries via the Consolidated HMS FMP resulted in the issuance of a non-jeopardy Opinion issued by NMFS on December 12, 2012. The Opinion included an ITS for loggerhead, Kemp's ridley, green, leatherback, and hawksbill sea turtles, all five Atlantic sturgeon DPSs, and smalltooth sawfish (See Table 18) (NMFS 2001a).

### *5.1.1.4 Atlantic Sea Scallop*

The Atlantic sea scallop fishery has a long history of operation in Mid-Atlantic and New England waters (NEFMC 1982, 2003). The fishery operates in areas and at times that it has traditionally operated and uses traditionally fished gear, which includes dredges and bottom trawls (NEFMC 1982, 2003). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2007). On Georges Bank and in the Mid-Atlantic, scallops are harvested primarily at depths of 30-100 meters, while the bulk of landings from the Gulf of Maine are from relatively shallow nearshore waters (<40 meters) (NEFSC 2007).

The Scallop FMP was originally implemented on May 15, 1982 (NEFSC 2007). Amendment 4 to the FMP, implemented in 1994, changed the management strategy from meat count regulation to effort control for the entire U.S. EEZ (NEFSC 2007). The limited access program, first established under Amendment 4, remains the basic effort control measure for the scallop fishery. From 2004 through 2008, vessels that did not qualify for a full-time, part-time, or occasional limited access permit could have obtained an open access, general category scallop permit. Effort (in terms of days fished) in the Mid-Atlantic is now about half of what it was prior to implementation of Amendment 4 to the Scallop FMP (NEFSC 2007).

An increase in active general category permits and landings from these vessels prompted the initiation of Amendment 11 to the Scallop FMP. In particular, it was noted that from 2000 to 2005 there was an increasing percentage of general category landings by vessels with homeports in the Mid-Atlantic region, and shifts in fishing effort by general category vessels to Mid-Atlantic fishing grounds (NEFMC 2007). In 2008, the implementation of Amendment 11 established a limited access general category program consisting of three permit types: Northern Gulf of Maine (NGOM), Incidental, and Individual Fishing Quota (IFQ). The IFQ program became effective March 1, 2010. The implementation of the LAGC fleet contributes to 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 observers as being captured in scallop dredge and or trawl gear. The first reported capture of a sea turtle in the scallop fishery occurred in 1996 during an observed trip of a scallop dredge vessel. A single capture in scallop dredge gear was reported for each of 1997 and 1999, as well. In 2001, 13 sea turtle captures in scallop dredge gear were observed and/or reported by NMFS 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 1996 that sea turtle interactions with scallop fishing gear occurred, there is no information to suggest that turtle interactions with scallop fishing gear are a new event or are occurring at a greater rate than what has likely occurred in the past. 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 1996. The average number of annual observable interactions of hard-shelled sea turtles in the Mid-Atlantic dredge fishery prior to the implementation of chain mats (January 1, 2001, through September 25, 2006) was estimated to be 288 turtles, of which 218 could be confirmed as loggerheads (Murray 2011). After the implementation of chain mats (September 26, 2006, through December 31, 2008), the average annual number of observable plus unobservable, quantifiable interactions in the Mid-Atlantic dredge fishery was estimated to be 125 turtles, of which 95 could be confirmed as loggerheads (Murray 2011). An estimate of loggerhead bycatch in Mid-Atlantic scallop trawl gear from 2005-2008 averaged 95 turtles annually (Warden 2011a). 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 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 17 or so years, has been present for decades.

Formal section 7 consultation on the continued authorization of the scallop fishery was last reinitiated on February 28, 2012, with an Opinion issued by NMFS on July 12, 2012. In this Opinion, NMFS determined that the continued authorization of the Scallop FMP (including the seasonal use of turtle deflector dredges [TDDs] in Mid-Atlantic waters starting in 2013) may adversely affect but was not likely to jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, and green sea turtles, or the five DPSs of Atlantic sturgeon, and issued an ITS (see **Error! Reference source not found.** below). The number of loggerhead and hard-shelled sea turtles expected to interact with scallop dredge gear annually is based on an analysis of sea turtle interactions in the dredge fishery from 2001-2008 as presented in Murray (2011). The number of loggerheads expected to interact with scallop trawl gear annually is based on data presented in Warden (2011a). For the other sea turtle species and Atlantic sturgeon, annual estimated interactions are based on observer data from the NEFOP and/or other bycatch reports. In the ITS, the scallop fishery is estimated to interact annually with up to 301 loggerhead, two leatherback, three Kemp’s ridley, and two green sea turtles, as well as one Atlantic sturgeon from any of the five DPSs. Of the loggerhead interactions, up to 112 per year are anticipated to be lethal from 2013 going forward. RPMs to minimize the impact of these incidental takes are also included in the Opinion, including an RPM to monitor fishing effort in the scallop dredge in the Mid-Atlantic during times when sea turtles are known to interact with the fishery (NMFS 2012c). Additional measures to minimize the impact of sea turtle interactions with the scallop fishery have been implemented through Frameworks 22 and 23 to the Scallop FMP and will be re-evaluated in future Frameworks.

#### *5.1.1.5 Atlantic Deep-Sea Red Sea Crab*

Section 7 consultation was completed on the red crab fishery during the proposed implementation of the Red Crab FMP (NMFS 2002b). The Opinion concluded that the action was not likely to result in jeopardy to any ESA-listed species under our jurisdiction. The fishery is a pot/trap fishery that occurs in deep waters along the continental slope. The primary fishing zone for red crab, as reported by the fishing industry, is at a depth of 1,300-2,600 feet along the continental shelf in the Northeast region, and is limited to waters north of 35°15.3'N (Cape Hatteras, NC) and south of the Hague Line. Following concerns that red crab could be overfished, an FMP was developed and became effective on October 21, 2002. In the 2002 Opinion, an ITS was provided for leatherback and loggerhead sea turtles (See Table 18 below).

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.

##### *5.1.1.6 Tilefish*

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 (9º-14ºC) 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 Opinion. The Opinion included an ITS for loggerhead and leatherback sea turtles (See Table 18 below) (NMFS 2001b).

##### *5.1.1.7 Shrimp Trawling in the Southeastern U.S.*

On December 2, 2002, our Southeast Regional Office (SERO) completed an Opinion for shrimp trawling in the southeastern U.S. on 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 existence of any sea turtle species (NMFS 2002a). This determination was based, in part, on the Opinion's analysis 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, SERO 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, 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 recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets., Fishing effort has been reduced by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007) and by about 40% in the South Atlantic (NMFS 2012a). As a result, sea turtle interactions and mortalities in the Gulf of Mexico, most notably for loggerheads and leatherbacks, were substantially less than projected in the 2002 Opinion.

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. The determination was based on elevated strandings suspected to be attributable to shrimp trawling, compliance concerns with TED and tow-time regulations, and elevated nearshore sea turtle abundance trawl catch per unit of effort (CPUE). These factors collectively indicated that sea turtles may be affected by shrimp trawling, under the sea turtle conservation regulations and federal FMPs, to an extent not considered in the 2002 opinion. The 2012 Opinion included an ITS for all five Atlantic sturgeon DPSs and smalltooth sawfish (See Table 18 below) (NMFS 2012a). Although the ITS in this Opinion did not provide actual estimates of incidental take for any sea turtle species, the effects section provided a qualitative assessment of likely impacts based on orders of magnitude (e.g., for Kemp's ridleys, at least tens of thousands and possibly hundreds of thousands of interactions are expected annually; of those interactions, thousands and possibly tens of thousands are expected to be lethal).

### 5.1.2 Federal Fisheries Included in This Opinion

The past effects of the seven federal fisheries currently being reviewed in this Opinion on ESA-listed species are discussed below.

#### *5.1.2.1 Northeast Multispecies*

The consultation history for the Northeast FMP appears above in section 2.1.1 and a description of this fishery appears above in section 3.2.

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, green, and leatherback sea turtles (NMFS 2001c). 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. The exact relationship between multispecies fishing effort and the number of endangered species interactions with gear used in the fishery is unknown. However, in general, less fishing effort results in less time that gear is in the water and therefore less opportunity for sea turtles or cetaceans to be captured or entangled in multispecies fishing gear.

In 2008, new information on the capture of loggerhead sea turtles in the NE multispecies fishery led to reinitiation of consultation. The October 29, 2010 Opinion concluded the continued operation of the multispecies 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 existence, and issued an ITS (See Table 18 below) (NMFS 2010d).

New information estimating loggerhead bycatch in bottom trawl gear has recently been published in Warden (2011). Using NEFOP data from 1996 to 2008 applied to VTR days fished, the average bycatch of loggerhead sea turtles in bottom otter trawl gear used in the NE multispecies fishery between 2005 and 2008 was estimated to be five loggerhead sea turtles per year (Warden 2011a). A thorough analysis of sea turtle interactions with gillnet and trawl gear is included in this consultation.

#### *5.1.2.2 Monkfish*

The consultation history for the Monkfish FMP appears above in section 2.1.2 and a description of this fishery appears above in section 3.3.

The directed monkfish fishery uses several gear types that may entangle protected species, including gillnet and trawl gear. Gillnet gear used in the monkfish fishery is known to capture ESA-listed sea turtles. Two unusually large stranding events occurred in April and May 2000, during which 280 sea turtles (275 loggerheads and five Kemp's ridleys) washed ashore on ocean facing beaches in North Carolina. Although there was not enough information to specifically determine the cause of the sea turtle deaths, there was information to suggest that the turtles died as a result of entanglement in gillnet gear, as four of the carcasses were carrying gillnet gear measuring 10-12 inches (24.5-30.5cm), which is consistent with gear used in the monkfish fishery. The monkfish and dogfish gillnet fisheries were both known to be operating in waters off North Carolina at the time the stranded turtles would have died. As a result, in March 2002, NMFS published new restrictions for the use of gillnets with larger than 8-inch (20.3 cm) stretched mesh in federal waters (3-200 nautical miles) off North Carolina and Virginia. In 2006, NMFS modified these requirements to apply to 7-inch (17.8 cm) stretched mesh in federal waters (3-200 nautical miles) off North Carolina and Virginia (71 FR 24776, April 26, 2006).

The most recent Opinion (October 29, 2010) concluded the continued operation of the monkfish 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 existence, and issued an ITS (See Table 18 below) (NMFS 2010c).

An estimate of loggerhead sea turtle bycatch in bottom otter trawl gear used in the monkfish fishery was published in a 2011 NEFSC Reference Document (Warden 2011a). Using NEFOP data from 1996 to 2008 applied to VTR days fished, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the monkfish fishery between 2005 and 2008 was estimated to be two loggerhead sea turtles per year (Warden 2011a).

#### *5.1.2.3 Spiny Dogfish*

The consultation history for the Spiny Dogfish FMP appears above in section 2.1.3, and a description of this fishery appears above in section 3.4.

The primary gear types for the spiny dogfish fishery are sink gillnets, otter trawls, bottom longline, and driftnet gear (NEFSC 2003). Recent data from fish dealer reports in FY 2008 indicate that spiny dogfish landings came mostly from sink gillnets (68.2%), and hook gear (15. 2%), bottom otter trawls (4.9%), as well as unspecified (7.7%) or other gear (3.9%) (MAFMC 2010). Sea turtles can be incidentally captured in all gear sectors of the spiny dogfish fishery, which can lead to injury and death as a result of forced submergence in the gear. ESA-listed cetaceans are also known to be seriously injured or killed from interaction with sink gillnet gear.

The most recent Opinion (October 29, 2010) concluded that operation of the spiny dogfish fishery may adversely affect ESA-listed sea turtles as a result of interactions with gillnet and trawl gear, and issued at ITS (See Table 18 below) (NMFS 2010f).

New information estimating loggerhead bycatch in bottom trawl gear has recently been published in Warden (2011). Using NEFOP data from 1996 to 2008 applied to VTR days fished, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the spiny dogfish fishery between 2005 and 2008 was estimated to be zero loggerhead sea turtles per year (Warden 2011a).

ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear and the cetacean distributions overlap. The

2010 Opinion concluded that the spiny dogfish fishery was not likely to jeopardize the existence of any ESA-listed species under our jurisdiction. Gillnet gear used in the spiny dogfish fishery is required to be in compliance with the ALWTRP.

#### *5.1.2.4 Atlantic Bluefish*

The consultation history for the Atlantic bluefish FMP appears above in section 2.1.4, and a description of this fishery appears above in section 3.5.

The primary gear types for the bluefish fishery are rod and reel, handline, pot, trap, and spear in the recreational fishery, which accounted for 72% of landing from 2004-2009 (MAFMC 2009). Gillnets and bottom otter trawl account for the majority of bluefish landed in the commercial fishery, and accounted for 97.1% of the total commercial directed catch and 79.6% of the total commercial trips targeting bluefish in 2008 (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)). The anticipated incidental take of ESA-listed sea turtles in bluefish fishing gear exempted by the 2010 Opinion (See Table 18 below) was based on observed interactions from sea sampling data for gear types targeting or capable of catching bluefish (NMFS 1999). The anticipated incidental take of loggerhead sea turtles was taken from the annual bycatch reports published by Murray (2006, 2008). At the time of the 2010 Opinion, the bluefish fishery was believed to interact with these species given the time and locations where the fishery occurred. Although no incidental takes of ESA-listed sea turtles had been reported in bottom otter trawl gear for trips that were ‘targeting’ bluefish (where greater than 50% of the catch was bluefish), incidental takes of loggerhead and Kemp’s ridley sea turtles were observed in bottom otter trawl gear where bluefish were caught but constituted less than 50% of the catch (NMFS 1999).

A new 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 (Warden 2011a). Using NEFOP data from 1996 to 2008 applied to VTR days fished, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the bluefish fishery between 2005 and 2008 was estimated to be four per year (Warden 2011a). The 2010 Opinion anticipated the annual incidental take of three loggerhead sea turtles. The trawl bycatch estimate described above represents new information on the effects of the bluefish fishery on ESA-listed sea turtles.

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

#### *5.1.2.5 Skate Complex*

The consultation history for the Skate Complex FMP appears above in section 2.1.5, and a description of this fishery appears above in section 3.6.

In 2010, bottom trawl gear accounted for 91.4% of directed skate landings. Gillnet gear is the next most common gear type, accounting for 8.3% of directed skate landings (NEFMC 2012). These numbers only refer to the skate bait fishery, and do not include landings of skate wings, which are usually caught incidentally in the multispecies and monkfish fisheries.The most recent (October 29, 2010) Opinion concluded that operation of the skate fishery may adversely affect ESA-listed sea turtles as a result of interactions with (capture in) gillnet and trawl gear, and issued an ITS (See Table 18 below) (NMFS 2010e). New information estimating loggerhead bycatch in bottom trawl gear has recently been published in Warden (2011). Using NEFOP data from 1996 to 2008 applied to VTR days fished, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the skate fishery for 2005-2008 was estimated to be seven loggerhead sea turtles per year (Warden 2011).

ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear and the cetacean distributions overlap. The 2010 Opinion concluded that the skate fishery was not likely to jeopardize the existence of any ESA-listed species under our jurisdiction. Gillnet gear used in the skate fishery is required to be in compliance with the ALWTRP.

#### *5.1.2.6 Mackerel/Squid/Butterfish*

The consultation history for the Mackerel, Squid, and Butterfish FMP appears above in section 2.1.6, and a description of this fishery appears above in section 3.7.

Gillnets account for a small amount of landings in the mackerel fishery, and all gillnet gear use by this fishery is subject to the requirements of the ALWTRP.

Loggerhead sea turtles are captured in bottom-otter trawl gear used in the *Loligo* and *Illex* squid fisheries, and gillnet gear used by the mackerel fishery and may be injured or killed as a result of forced submergence in the gear. The most recent (October 29, 2010) Opinion concluded that the continued operation of the fishery under the FMP was likely to adversely affect sea turtles, but not jeopardize their continued existence. An ITS was provided with the 2010 Opinion along with non-discretionary RPMs to minimize the impacts of incidental take (See 16 below).

### *5.1.2.7 Summer Flounder/Scup/Black Sea Bass*

The consultation history for the Summer Flounder, Scup, and Black Sea Bass FMP appears above in section 2.1.7 and a description of this fishery appears above in section 3.8.

In 2006, the NEFSC released an estimate of loggerhead sea turtle takes in bottom otter trawl gear fished in Mid-Atlantic waters during the period 1996-2004 (Murray 2006). Fifty percent of the observed 66 takes occurred on vessels targeting summer flounder. However, it should also be noted that some of the observed interactions occurred on vessels fishing with TEDs using an allowed (at that time) TED extension with a minimum 5.5-inch mesh (Murray 2006). Numerous problems were documented 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-inch 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, NC, and seasonally (March 16-January 14) for trawl vessels fishing between Oregon Inlet, NC, and Cape Charles, VA. Effort in the summer flounder, scup, and black sea bass fisheries has also declined since the 1980s and since each fishery became managed under the FMP. Therefore, effects to sea turtles are expected, in general, to have declined as a result of the decline in fishing effort. Nevertheless, the fisheries primarily operate in Mid-Atlantic waters in areas and times when sea turtles occur. Thus, there is a continued risk of sea turtle captures causing injury and death in summer flounder, scup, and black sea bass fishing gear.

The most recent section 7 consultation (October 29, 2010) concluded that operation of the summer flounder, scup and black sea bass fishery may adversely affect ESA-listed whales and sea turtles as a result of interactions with (capture in) trawl, gillnet, and pot/trap gear, and issued an ITS (See 17 below) (NMFS 2010g).

New information estimating loggerhead bycatch in bottom trawl gear has recently been published in Warden (2011). Using NEFOP data from 1996 to 2008 applied to VTR days fished, the average bycatch of loggerhead sea turtles in bottom otter trawl gear used in the summer flounder, scup, and black sea bass fishery between 2005 and 2008 was estimated to be 110 loggerhead sea turtles per year (Warden 2011).

ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear and the cetacean distributions overlap. The 2010 Opinion concluded that the summer flounder, scup, and black sea bass fishery was not likely to jeopardize the continued existence of any ESA-listed species under our jurisdiction. Gillnet gear used in the summer flounder, scup, and black sea bass fishery is required to be in compliance with the ALWTRP.

#### *5.1.2.8 Summary of ITSs in Federal Fishery Opinions*

**Table 18:** Dates of the most recent Opinions regarding federal fisheries prepared by NMFS NERO and SERO for federally managed fisheries in the action area and their respective ITSs for sea turtles. Unless noted, levels of incidental take exempted are on an annual basis.

| FMP | Date of Most Recent Opinion | Loggerhead | Kemp's Ridley | Green | Leatherback |
|---|---|---|---|---|---|
| American lobster | August 10, 2012 | 1 | 0 | 0 | 5 |
| Atlantic sea scallop | July 12, 2012 | 301 (112 lethal from 2013 on) | 3 | 2 | 2 |
| Atlantic bluefish | October 29, 2010 | 82 (34 lethal) | 4 | 5 | 4 |
| Monkfish | October 29, 2010 | 173 (70 lethal) | 4 | 5 | 4 |
| Multispecies | October 29, 2010 | 46 (21 lethal) | 4 | 5 | 4 |
| Skate | October 29, 2010 | 39 (17 lethal) | 4 | 5 | 4 |
| Spiny dogfish | October 29, 2010 | 2 | 4 | 5 | 4 |
| Mackerel/squid/ butterfish | October 29, 2010 | 62 (25 lethal) | 2 | 2 | 2 |
| Summer flounder/scup/ black sea bass | October 29, 2010 | 205 (85 lethal) | 4 | 5 | 6 |
| Shark fisheries under the Consolidated HMS FMP | December 12, 2012 | 126 (78 lethal) every 3 years | 36 (21 lethal) every 3 years | 57 (33 lethal) every 3 years | 18 (9 lethal) every 3 years |
| Coastal migratory pelagics (mackerel) | August 13, 2007 | 33 every 3 years | 4 every 3 years | 14 every 3 years | 2 every 3 years |
| Red Crab | February 6, 2002 | 1 | 0 | 0 | 1 |
| South Atlantic snapper-grouper | June 7, 2006 | 202 (67 lethal) every 3 years | 19 (8 lethal) every 3 years | 39 (14 lethal) every 3 years | 25 (15 lethal) every 3 years |
| Pelagic longline fishery HMS FMP (per the RPA) | June 1, 2004 | 1,905 (339 lethal) every 3 years | *105 (18 lethal) every 3 years | *105 (18 lethal) every 3 years | 1764 (252 lethal) every 3 years |
| South-Atlantic dolphin-wahoo** | August 27, 2003 | 12 (2 lethal) every 3 years | 2 (1 lethal) every 3 years | 2 (1 lethal) every 3 years | 12 (1 lethal) every 3 years |
| Southeastern shrimp trawling*** | May 8, 2012 | Not able to be estimated | Not able to be estimated | Not able to be estimated | Not able to be estimated |
| Tilefish | March 13, 2001 | 6 (3 lethal) | 0 | 0 | 1 |

**combination of 105 (18 lethal) Kemp's ridley, green, hawksbill, or olive ridley*
***combination of 16 turtles total every three years with two lethal (Kemp's ridley, green, hawksbill, leatherback)*
**** although the ITS in this Opinion does not provide actual estimates of incidental take for any sea turtle species, the effects section provides a qualitative assessment of likely impacts based on orders of magnitude, estimating that the shrimp fishery, as currently operating, would result in at least*

*thousands and possibly tens of thousands of interactions annually, of which at least hundreds and possibly thousands are expected to be lethal (NMFS 2012a).*

**Table 19** Opinions regarding federal fisheries prepared by NMFS NERO and SERO for federally managed fisheries in the action area and their respective ITSs for Atlantic sturgeon since the ESA listing of Atlantic sturgeon. Unless noted, levels of incidental take exempted are on an annual basis.

| FMP | Date of Most Recent Opinion | GOM DPS | NYB DPS | CB DPS | Carolina DPS | SA DPS |
|---|---|---|---|---|---|---|
| American lobster | August 10, 2012 | none | | | | |
| Atlantic sea scallop* | July 12, 2012 | 1 | 1 | 1 | 1 | 1 |
| Shark fisheries under the Consolidated HMS FMP | December 12, 2012 | 36 over 3 years with 9 being lethal take | 159 over 3 years with 30 being lethal take | 45 over 3 years with 9 being lethal take | 18 over 3 years with 6 being lethal take | 63 over 3 years with 12 being lethal take |
| Southeastern shrimp trawling | May 8, 2012 | 156 interactions over 3 years (24 captures, 3 lethal) | 450 interactions over 3 years (66 captures, 9 lethal) | 312 interactions over 3 years (48 captures, 6 lethal) | 498 interactions over 3 years (75 captures, 9 lethal) | 1356 interactions over 3 years (198 captures, 24 lethal) |

**1 take annually in the scallop trawl fishery from any of the 5 DPSs; 1 lethal take over 20 years*

### 5.1.3 Hopper Dredging

The construction and maintenance of federal navigation channels and sand mining ("borrow") areas have also been identified as sources of sea turtle mortality. Atlantic sturgeon may also be killed during hopper dredging operations, although this is rare. All hopper dredging projects are authorized or carried out by the U.S. Army Corps of Engineers (Corps). In the northern portion of the action area, these projects are under the jurisdiction of the districts within the North Atlantic Division or the Wilmington District. Hopper dredging projects in this area have resulted in the recorded mortality of approximately 87 loggerheads, four greens, nine Kemp's ridleys and four unidentified hard shell turtles since observer records began in 1993. To date, nearly all of these interactions have occurred in nearshore coastal waters with very few interactions in the open ocean. Few interactions between hopper dredges and Atlantic sturgeon have been reported, with just three records documenting interactions between hopper dredges and Atlantic sturgeon in the action area (two in Virginia near the Chesapeake Bay entrance, and one in New York Bight). We and the Southeast region have completed several ESA section 7 consultations with the Corps to consider effects of these hopper dredging projects on listed sea turtles. Many of these consultations will be reinitiated to consider effects to Atlantic sturgeon. Table 20 below provides information on Opinions considering dredging projects in the action area and the associated ITS for sea turtles (unless otherwise noted, take estimates are per dredge cycle):

**Table 20** Information on consultations conducted by NMFS for dredging projects that occur in Maine through North Carolina (See below for a separate explanation of consultations on projects from South Carolina through Florida that overlap with the action area).

| Project | Date of Opinion | Loggerhead | Kemp's ridley | Green | Leatherback | Notes |
|---|---|---|---|---|---|---|
| USCOE - Continued Hopper Dredging of Channels & Borrow Areas in SE U.S. | 9/25/1997 | 24 | 7 | 7 | 0 | Annual Estimate |
| Dredging of Sandbridge Shoals, VA | 4/2/1993 | 5 | 1 Kemp's ridley or green | | 0 | |
| Long Island NY to Manasquan NJ Beach Nourishment | 12/15/1995 | 5 turtles total: combination of any species | | | | |
| Sandy Hook Channel Dredging | 6/10/1996 | 2 | 1 | 2 | 1 | 2 loggerheads/ green inclusive; and 1 Kemp's/leatherback |
| ACOE Philadelphia District Dredging | 11/26/1996 | 4 | 1 | 1 | 0 | Annual Estimate |

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

Since 1991, our Southeast Regional Office (SERO) has issued three regional biological opinions (RBOs) regarding Army Corps of Engineers (ACOE) hopper dredging in the South Atlantic District. Most recently, in September 1997, SERO issued an RBO on *The continued hopper dredging of channels and borrow areas in* *the southeastern United States*, authorizing the take of threatened and endangered species by ACOE dredging activities in the South Atlantic District.

To date, use of hopper dredges in ACOE activities in northeast Florida and Georgia has been limited under the 1997 RBO to operating between December 1 through April 15, except in emergency situations, and the dredging projects have had to abide by the reasonable and prudent measures, and terms and conditions set forth in the 1997 RBO. Federal actions that are consistent with the RBO fall under its ITS, which set an annual documented incidental take for the region of seven Kemp's ridley, seven green, two hawksbill, and 35 loggerhead sea turtles. Other federal actions that are not within the scope of the RBO have undergone separate consultations, for which we issued Opinions and Incidental Take Statements.

#### 5.1.4 Research and Other Permitted Activities

Research activities either conducted or funded by federal agencies within the action area may adversely affect ESA-listed marine mammals, sea turtles, and fish, and may require a section 7 consultation. Several section 7 consultations on research activities have recently been completed, as described below:

*NEFSC Fisheries Surveys*
NOAA research vessels conducting fisheries surveys for the NEFSC are estimated to capture no more than 11 sea turtles and nine Atlantic sturgeon per year, primarily using trawl gear. This includes up to seven NWA DPS loggerheads, one leatherback, two Kemp's ridleys, and one green sea turtle, as well as four NYB, two SA, one GOM, one CB, and one Carolina DPS origin Atlantic sturgeon per year (NMFS 2012d). With the exception of one loggerhead and one Kemp's ridley, none of these sea turtles or Atlantic sturgeon are expected to die, immediately or later, as a result of capture in the sampling gear.

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

*Long Island Sound Trawl Survey*
The Long Island Sound Trawl Survey (LISTS) conducted by the State of Connecticut Department of Energy and Environmental Protection, Marine Fisheries Division, (CT DEEP) are also expected to result in incidental takes of ESA-listed species, including two sea turtles (one Kemp's ridley, green or leatherback, and one NWA DPS loggerhead), and a total of no more than 120 Atlantic sturgeon, of which 94 will be NYB DPS origin, 12 will be SA DPS origin, eight will be CB DPS origin, five will be GOM DPS origin and one will be Carolina DPS origin (NMFS 2012f). No mortalities of any ESA-listed species are expected.

*State of New Jersey Marine Surveys*
The marine surveys carried out by the State of New Jersey under the Dingell-Johnson Sport Fish Restoration Funding program are expected to take five NWA DPS loggerhead sea turtles, one Kemp's ridley, green, or leatherback sea turtle; and a total of no more than 109 Atlantic sturgeon (62 NYB DPS, 20 CB DPS, 19 SA DPS and eight GOM DPS origin) (NMFS 2012g). No mortalities of any ESA-listed species are expected.

*Section 10 Permits*
NMFS has issued additional research permits under section 10(a)(1)(A) of the ESA, which authorizes activities for scientific purposes or to enhance the propagation or survival of the affected species. The permitted activities do not operate to the disadvantage of the species and are consistent with the purposes of the ESA, as outlined in section 2 of the Act. The following section 10(a)(1)(A) permits are currently in effect for sea turtles (Table 21) and Atlantic sturgeon (Table 22). No section 10 permits that authorize serious injury or mortality of marine mammals are currently in effect.

**Table 21:** Active section 10 permits authorizing take of sea turtles for scientific research

| Permittee | File # | Project | Area | Sea Turtle Takes | Dates |
|---|---|---|---|---|---|
| Coonamessett Farm Foundation, Inc, Research, East Falmouth, MA | 14249 | Sea Turtle- Scallop Dredge Interaction Studies | Atlantic Ocean DE,MD,NC, NJ,NY,VA | 17 loggerheads captured by dredges, 10 for satellite tagging or crittercam, 100 followed by ROVs.<br>6 of any other species (Kemp's, green, leatherback, olive ridley, hawksbill) | 01/16/2009-10/31/2014 |
| South Carolina Department of Natural Resources | 15566 | Assessing change in distribution and health of sea turtles in coastal waters between Winyah Bay, SC and St. Augustine, FL | Coastal Waters b/t Winyah Bay, SC and St. Augustine, FL | 345 loggerheads, 79 Kemp's ridley, 9 green, 1 leatherback, 1 hawksbill<br>Unintentional mortalities: 5 loggerhead, 1 Kemp's ridley, 1 green, 1 leatherback, 1 hawksbill over course of permit | 04/08/2011-04/30/2016 |
| NMFS Northeast Fisheries Science Center (NEFSC) | 1576 | PR1 Permit #1576 scientific research | Projects 1,2,3: Western Atlantic Ocean (Maine through the Florida Keys); Project 4: (Gulf of Maine through North Carolina) | Project 1: 23 loggerheads, 1 leatherback, 1 Kemp's ridley, 1 green; scallop dredge research without chain mats-could result in all lethal takes.<br>Project 2: 50 loggerheads, 7 leatherbacks, 25 Kemp's ridleys, 9 greens (all takes authorized under ITS)<br>Project 3: 8 loggerheads, 2 leatherbacks, 1 Kemp's ridley, 1 green, 1 hawksbill, 1 olive ridley (Capture | 11/02/2006-10/31/2012 |

| | | | | | |
|---|---|---|---|---|---|
| | | | | authorized under Apex predator project)<br>Project 4: 50 loggerheads, 1 mortality over course of permit; 50 Kemp's ridley, 1 mortality over course of permit; 50 unidentified (necropsy/salvage) | |

**Table 22:** Active section 10 permits authorizing take of Atlantic sturgeon for scientific research

| Permittee | File # | Project | Area | Atlantic Sturgeon Takes | Dates |
|---|---|---|---|---|---|
| New York State Department of Environmental Conservation | 16436 | Section 10 permit for research and monitoring of Atlantic sturgeon in the Hudson River Estuary | Hudson River | Miles 25-43 Hudson River: 300 juveniles<br>Miles 60-115 Hudson River: 200 adults<br>Miles 25-115 Hudson River: 1050 juveniles<br>Up to 2 annual mortalities or harm of juvenile fish (<1,000mm) | 04/04/2012-04/05/2017 |
| Environmental Research and Consulting, Kennett Square, PA | 16438 | Scientific research on Atlantic sturgeon in the Delaware River and Bay | Delaware River and Bay / Mouth of Delaware Bay | 300 juveniles, up to 5 serious injuries/mortalities over course of permit | 04/04/2012-04/05/2017 |
| University of Georgia | 16482 | Population Dynamics and Seasonal Habitat Use of Atlantic sturgeon in Georgia | Savannah River | 3,700 individuals annually, up to 5 unintentional mortalities in all rivers annually | 04/04/2012-04/05/2017 |
| Maine Department of Marine Resources (MDMR) | 16526 | Atlantic sturgeon of the Gulf of Maine | Penobscot, Kennebec, Saco, Merrimack and small coastal rivers | 1175 individuals annually; Incidental mortality of up to 3 over five years, but up to one adult or subadult | 04/04/2012-04/05/2017 |
| Connecticut Department of Environmental Protection | 16323 | Monitor Atlantic Sturgeon in CT waters | Connecticut, Thames, and Housatonic rivers, Long Island Sound | 200 adults and subadults per year, 0 incidental mortality | 04/04/2012-04/05/2017 |
| Delaware Division of Fish and Wildlife | 16431 | Locate juvenile Atlantic sturgeon nursery habitat, assess movement panems and habitat use in the Delaware River. | Delaware River, New York Bight | 240 juveniles per year, 1 incidental mortality | 04/04/2012-04/05/2017 |
| NOAA Fisheries Northeast Region, Protected Resources Division | 1614-04 | Maximize the use ofdead Atlantic and shortnose sturgeon for research and educational purposes | Eastern coast and rivers of the US in NER and SER | 450 shortnose sturgeon, 175 Atlantic sturgeon per year (dead animals only) | 5/30/201-2/28/2013 |
| Stony Brook University | 16422 | Examine movements of Atlantic srurgeon within oceanic habitat using an offshore bottom trawl survey | Coast of Connecticut, New York, New Jersey and Delaware | 325 adults and subadults per year, 0 incidental mortality | 04/04/2012-04/05/2017 |
| U.S. Fish and Wildlife Service | 16547 | Atlantic sturgeon research in the Chesapeake Bay | Chesapeake Bay | 425, but no more than 150 adults and 75 juveniles captured in any one river per year. Also, no more than 75; Total of 3 annual mortalities or harm of Atlantic sturgeon from all areas of research | 04/04/2012-04/05/2017 |
| Delaware State University | 16507 | Study adult Atlantic sturgeon | Delaware bay, rivers, and | 350 eggs/larvae per year, 100 juveniles per year, 410 adults/subadults per year; | 04/04/2012-04/05/2017 |

| | | | | | |
|---|---|---|---|---|---|
| | | abundance, distribution, movement, habitat use, and spawning | coastal waters | intentional harvesting of eggs, 0 incidental mortality | |

Scientific research on ESA-listed Atlantic salmon has been authorized under the USFWS' endangered species blanket permit (No. 697823) under section 10(a)(1)(A), and covers a number of research projects carried out by NMFS and other research partners contracted by NMFS such as the University of Maine. The USFWS is anticipating re-structuring their permits in 2013 and will issue new permits to cover only research directly under the direct supervision of NMFS and will no longer be providing authorization (i.e., sub-permits) for research being conducted by other entities. The USFWS is planning to issue separate permits for different research activities conducted through other agencies or partners such as USGS, Maine DMR, University of Maine. This will provide a more efficient way of tracking individual take and will allow the USFWS to have a better understanding of ongoing research and level of take associated with these activities through the annual reporting requirements.

NOAA Fisheries currently cooperates in research on Atlantic salmon in the Penobscot River to document changes in fish populations resulting from both the removal of the Veazie and Great Works Projects as well as the construction of the fish bypass at the Howland Project. The study uses boat electrofishing techniques to document baseline conditions in the river prior to construction at the dams. Following dam removal and construction of the fish bypass, researchers will re-sample the river.

NOAA Fisheries also is monitoring biomass and species composition in the estuary to look at system-wide effects of dame removal projects. Although these activities will result in some take of Atlantic salmon, adverse impacts are expected to be minor and authorized by the existing ESA permit. The information gained from these activities will be used to further salmon conservation actions in the GOM DPS.

USFWS is authorized to conduct the conservation hatchery program at the Craig Brook and Green Lake National Fish Hatcheries. The mission of the hatcheries is to raise Atlantic salmon parr and smolts for stocking into selected Atlantic salmon rivers in Maine. Over 90% of adult returns to the GOM DPS are currently provided through production at the hatcheries. Approximately 600,000 smolts are stocked annually in the Penobscot River. The hatcheries provide a significant buffer from extinction for the species.

*Section 10(a)(1)(B) Permits*

Section 10(a)(1)(B) of the ESA authorizes NMFS, under some circumstances, to permit non-federal parties to take otherwise prohibited fish and wildlife if such taking is "incidental to, and not the purpose of carrying out otherwise lawful activities" (50 CFR 217-222). As a condition for issuance of a permit, the permit applicant must develop a conservation plan that minimizes negative impacts to the species. There are currently three active Section 10(a)(1)(B) permits (Table 23):

**Table 23** Active Section 10(a)(1)(B) permits

| Permittee | File # | Project | Area | Annual Endangered Species Takes | Dates |
|---|---|---|---|---|---|
| Georgia Department of Natural Resources | 16645 | Commercial shad fishery conservation plan | Atlamaha River, Savannah River, Ogeechee River | Atlamaha: 140 Atlantic sturgeon (2.3% mortality)<br>Savannah: 50 Atlantic sturgeon (2.3% mortality)<br>Ogeechee: 10 Atlantic sturgeon (2.3% mortality) | 2013-2022 |
| Virginia Polytechnic Institute | 1529 | Annual horseshoe crab abundance monitoring surveys | State and federal waters of Cape Cod, MA, to the GA-FL border | Leatherback: 1 live or dead;<br>Loggerhead: 34 live, 2 dead;<br>Kemp's ridley: 14 live, 1 dead;<br>Green 2 live, 1 dead. | 2005-present |
| North Carolina Division of Marine Fisheries | 1528 | Large and small mesh gillnet fishing | Pamlico Sound, NC | Leatherback: 2 live or dead;<br>Loggerhead: 38 live, 3 dead;<br>Kemp's ridley: 27 live, 14 dead;<br>Green 120 live, 48 dead;<br>Hawksbill: 2 live or dead | 2005-present |

In addition, most coastal Atlantic states are either in the process of applying for permits or considering applications for state fisheries. Active permits and permit applications are posted online for all species as they become available at http://www.nmfs.noaa.gov/pr/permits/esa_review.htm. We are actively working with several states and other parties on section 10(a)(1)(B) permits; however to date no section 10(a)(1)(B) permits have been authorized for GOM DPS Atlantic salmon or ESA-listed cetaceans.

*MMPA Incidental Harassment Authorizations and Letters of Authorization*
Under Section 101(a)(5) of the MMPA, certain incidental taking of a small number of marine mammals by U.S. citizens who are engaged in an activity other than commercial fishing is allowed through the issuance of Incidental Harassment Authorizations (IHAs) or Letters of Authorization (LOAs). IHAs allow applicants to use an expedited process (4-8 months) for authorization to incidentally "harass" marine mammals as long as there is no potential for serious injury/mortality or the potential for serious injury/mortality can be negated through mitigation measures that could be required under the authorization. If the potential for serious injury/mortality exists and no mitigating measures can be taken to prevent this kind of take, than the applicant must apply for an LOA. The LOA process takes 8-18 months.

The types of activities receiving IHAs and LOAs may involve acoustic harassment or habitat disturbance from yacht races (America's cup), seismic surveys, exploratory drilling surveys, bridge construction, fireworks displays, sonar testing, Navy training exercises, and light house restorations, among others. The types of authorized takes include behavioral responses, as well as injuries and mortalities. Currently there are no LOAs that allow serious injuries and mortalities for ESA-listed cetaceans. Current and past applications are available for public review at http://www.nmfs.noaa.gov/pr/permits/incidental.htm#applications. NMFS performs section 7 consultations on the issuance of IHAs and LOAs that may affect listed species.

### 5.1.5 Vessel Activity and Military Operations

Potential sources of adverse effects to ESA-listed species 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 Bureau of Ocean Energy Management (BOEM), Federal Energy Regulatory Commission (FERC), and Maritime Administration (MARAD) on vessel traffic related to energy projects in the Northeast Region and has implemented conservation measures. Through the section 7 process, where applicable, NMFS has and will continue to identify conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species.

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

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

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

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

In addition to section 7 consultations, our Headquarters Office issues Incidental Harassment Authorizations (IHAs) and Letters of Authorization (LHAs) under the MMPA that allow the U.S. Armed Forces to harass a certain number of marine mammals in the course of their operations. The harassments authorized do not rise to the level of serious injuries or mortalities, and so are not considered further in this Opinion.

### 5.2 Non-Federally Regulated Fisheries

Several fisheries for species that are not managed by a federal FMP occur in the action area. The amount of gear these fisheries use is unknown. In most cases, there is limited observer coverage of these fisheries and the extent of interactions with ESA-listed species is unknown.

Atlantic sturgeon, Atlantic salmon, cetaceans, and sea turtles may be vulnerable to capture, injury, and mortality in fisheries occurring in state waters. Captures of Atlantic sturgeon (ASSRT 2007; NMFS 2011a) and sea turtles in nearshore fisheries have been reported (NMFS SEFSC 2001; ASMFC 2006; ASMFC 2007). Bycatch of Atlantic salmon in state recreational and commercial fisheries have also been reported, but little quantitative data exist that would allow meaningful estimation of their effects (AASBRT 2006). After the closure of Atlantic salmon fisheries, some poaching and misidentification has been documented. Area closures and a 25-inch maximum length regulation have been put in place in Maine to protect Atlantic salmon. Federal enforcement officials and the Maine Warden Service work together on Atlantic salmon surveillance and poaching investigations. Atlantic salmon are unlikely to be present in other state fisheries, and are not discussed further in this section.

Information on the number of Atlantic sturgeon captured or killed in state water fisheries is extremely limited. Efforts are currently underway to obtain more information on the numbers of Atlantic sturgeon captured and killed in state water fisheries. Atlantic sturgeon are vulnerable to capture in state fisheries occurring in rivers, such as shad fisheries; however, these riverine areas are outside the action area under consideration in this Opinion. Where available, specific information on sea turtle and sturgeon interactions in state fisheries is provided below.

The available bycatch data for FMP fisheries indicate that sink gillnets and otter trawl gear pose the greatest risk to Atlantic sturgeon (ASMFC 2007), although Atlantic sturgeon occasionally are caught by hook and line, fyke nets, and crab pots (NMFS 2011a).

Observations of state recreational fisheries have shown that loggerhead, leatherback, Kemp's ridleys 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 (NMFS SEFSC 2001). A summary of known impacts of hook-and-line captures on loggerhead sea turtles can be found in the TEWG (1998, 2000, 2009) reports.

Atlantic sturgeon have been observed captured in hook and line gear; the number of interactions that occur is unknown. While most Atlantic sturgeon are likely to be released alive, we currently have no information on post-release survival. Information in this section is presented by fishery, with state-specific information where available.

### 5.2.1 Atlantic Croaker

An Atlantic croaker fishery using trawl and gillnet gear occurs within the action area and turtle takes have been observed in the fishery. The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the Atlantic croaker fishery was estimated to be 70 loggerhead sea turtles (Warden 2011). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the Atlantic croaker fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the Atlantic croaker fishery, based on VTR data from 2002 to 2006, was estimated to be 11 per year with a 95% CI of 3-20 (Murray 2009b). ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear overlaps with cetacean distributions.

Atlantic sturgeon takes have been observed in the Atlantic croaker fishery, but a quantitative assessment of the number of Atlantic sturgeon captured in the croaker fishery is not available. A review of the NEFOP database indicates that, from 2006 to 2010, 60 Atlantic sturgeon (out of a total of 726 observed interactions) were captured during observed trips where the trip target was identified as croaker. This represents a minimum number of Atlantic sturgeon captured in the croaker fishery during this time period, as it only considers trips that included a NEFOP observer onboard. It should also be noted that very few croaker trips carry NEFOP observers.

### 5.2.2 Weakfish

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 gillnet landings began to account for most weakfish landed (ASMFC 2002). North Carolina has accounted for the majority of the annual landings since 1972 while Virginia ranks second, followed by New Jersey (ASMFC 2002). Sea turtle bycatch in the weakfish fishery has occurred (Warden 2011; Murray 2009a, 2009b). The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the weakfish fishery was estimated to be one loggerhead sea turtle (Warden 2011). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the weakfish fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the weakfish fishery, based on VTR data from 2002 to 2006, was estimated to be one per year with a 95% CI of 0-1 (Murray 2009b). ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear overlaps with cetacean distributions.

A quantitative assessment of the number of Atlantic sturgeon captured in the weakfish fishery is not available. Mortality rates of Atlantic sturgeon in commercial trawls has been estimated at 5%. A review of the NEFOP database indicates that from 2006 to 2010, 36 Atlantic sturgeon (out of a total of 726 observed interactions) were captured during observed trips where the trip target was identified as weakfish. This represents a minimum number of Atlantic sturgeon captured in the weakfish fishery during this time period, as it only considers observed trips, and most inshore fisheries are not observed. An earlier review of bycatch rates and landings for the weakfish fishery reported that the weakfish-striped bass fishery had an Atlantic sturgeon bycatch rate of 16% from 1989 to 2000; the weakfish-Atlantic croaker fishery had an Atlantic sturgeon bycatch rate of .02%,[30] and the weakfish fishery had an Atlantic sturgeon bycatch rate of 1.0% (ASSRT 2007).

### 5.2.3 Whelk

A whelk fishery using pot/trap gear is known to occur in several parts of the action area, including waters off Maine, Connecticut, Massachusetts, Delaware, Maryland, and Virginia. Landings data for Delaware suggests that the greatest effort in the whelk fishery for its waters occurs in the months of July and October, times when sea turtles are present. Whelk pots, which unlike lobster traps are not fully enclosed

[30] Bycatch rates were calculated as pounds of sturgeon per pound landed (Stein *et al*. 2004a)

and differ in use of a bridle, 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, green, and loggerhead sea turtles as well as right, humpback, and fin whales are known to become entangled in lines associated with trap/pot gear used in several fisheries including lobster, whelk, and crab species (NMFS SEFSC 2001; Dwyer *et al.* 2002; NMFS 2007a). Atlantic sturgeon are not known to interact with whelk pots.

#### 5.2.4 Crab

Various crab fisheries**,** such as horseshoe crab and blue crab, also occur in federal and state waters. Leatherback, green, and loggerhead sea turtles as well as right, humpback, and fin whales are known to become entangled in lines associated with trap/pot gear used in several fisheries including lobster, whelk, and crab species (NMFS SEFSC 2001; Dwyer *et al.* 2002: NMFS 2007a).

The crab fisheries may have detrimental impacts on sea turtles beyond entanglement in the fishing gear itself. Loggerheads are known to prey on crab species, including horseshoe and blue crabs. In a study of the diet of loggerhead sea turtles in Virginia waters from 1983 to 2002, Seney and Musick (2007) found a shift in the diet of loggerheads in the area from horseshoe and blue crabs to fish, particularly menhaden and Atlantic croaker. The authors suggested that a decline in the crab species has caused the dietary 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 of 2003, declines were evident in some parts of the Mid-Atlantic (ASMFC 2004; Eyler *et al.* 2007). Given the variety of loggerheads prey items (Dodd 1988; Burke *et al.* 1993; Bjorndal 1997; Morreale and Standora 1998) and the differences in regional abundance of horseshoe crabs and other prey items (ASMFC 2004; Eyler *et al.* 2007), a direct correlation between loggerhead sea turtle abundance and horseshoe crab and blue crab availability cannot be made at this time. Nevertheless, the decline in loggerhead abundance in Virginia waters (Mansfield 2006), and possibly Long Island waters (Morreale *et al.* 2005), coincident with noted declines in the abundance of horseshoe crab and other crab species raises concerns that crab fisheries may be impacting the forage base for loggerheads in some areas of their range.

Atlantic sturgeon are known to be caught in state water horseshoe crab fisheries (Stein *et al*. 2004a), which currently operate in all action area states except New Jersey. Along the East Coast, hand, trawl, and dredge fisheries account for more than 85% of the commercial horseshoe crab landings in the bait fishery. Other methods used are gillnets, pound nets, and traps (ASMFC 2011). State waters from Delaware to Virginia are closed to horseshoe crab harvest and landing from January

1 to June 7 (ASMFC 2011). The majority of horseshoe crab landings in 2010 came from Massachusetts, Virginia, and Delaware. Stein *et al.* (2004a) examined bycatch of Atlantic sturgeon using the NEFOP database (1989-2000) and found that their bycatch rate in horseshoe crab fisheries was low, at 0.05%.

An Atlantic sturgeon "reward program"—where commercial fishermen were provided monetary rewards for reporting captures of Atlantic sturgeon in the Maryland waters of Chesapeake Bay—operated from 1996 to 2012 (Mangold *et al.* 2007).[31] The data from this program during the ten-year period of 1996-2006 show that of the 1,395 wild Atlantic sturgeon, only one was found caught in a crab pot (Mangold *et al.* 2007).

#### 5.2.5 American Lobster Trap and Fish Trap Fisheries

An American lobster trap fishery occurs in state waters of New England and the Mid-Atlantic and is managed under the ASMFC's Interstate Fishery Management Plan (ISFMP). As with the federal waters component of the fishery, the state waters fishery is known to have the potential to entangle leatherback and loggerhead sea turtles as well as right, humpback, and fin whales in lines associated with trap/pot gear used in this fishery (NMFS SEFSC 2001; Dwyer *et al.* 2002; NMFS 2007a).

The American lobster fishery has been verified as the gear/fishery involved in 43 leatherback entanglements in the Northeast Region between 2002 and 2010 (STDN 2012). All of the 43 entanglements involved vertical line of the gear. These probable/confirmed entanglements have occurred in ME, MA, RI, and one in CT. These entanglements have occurred from May through October. Gear has been verified through the buoy/gear identification numbers, which can be traced in the various state agency and federal permit systems. Of the 43 confirmed or probable sets of gear, one has been verified as MA recreational lobster pot gear (entangled a leatherback in August 2006), and two sets of gear have been identified to a fisherman with both MA State and federal permits for lobster pot gear. Four of the entanglements involved gear from fishermen with state permits, and possibly federal permits, but this could not be confirmed. In seven of the entanglements, it was unknown if the gear came from a state, federal, or recreational fishery. All other lobster gear has been confirmed to be state commercial (ME, MA, CT or RI) coastal lobster pot gear.

Bycatch of loggerheads in fish traps have also been reported from several Atlantic coast states (Shoop and Ruckdeschel 1989; W. Teas, pers. comm.). No information on interactions between Atlantic sturgeon and fish traps, long haul seines, or channel nets is currently available; however, depending on where this gear is set and the mesh size, the potential exists for Atlantic sturgeon to be entangled or captured in this gear.

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

### 5.2.6 Northern Shrimp

A Northern shrimp fishery also occurs in state waters of Maine, New Hampshire, and Massachusetts, and is managed under the ASMFC's ISFMP. In 2010, the ISFMP implemented a 126-day season, from December 1 to April 15, but the shrimp fishery has exceeded its TAC and closed early every year, ending on February 17 in 2012. The majority of northern shrimp are caught with otter trawls, which must be equipped with Nordmore grates (ASMFC NSTC 2011). Otter trawls in this fishery are known to interact with Atlantic sturgeon, but exact numbers are not available (NMFS 2011a). A significant majority (84%) of Atlantic sturgeon bycatch in otter trawls occurs at depths <20 meters, with 90% occurring at depths of <30 meters (ASMFC 2007). During the spring and fall inshore trawl surveys, northern shrimp are most commonly found in tows with depths of > 64 meters (ASFMC NSTC 2011), which is well below the depths at which most Atlantic sturgeon bycatch is occurring. Atlantic sturgeon are known to interact with shrimp trawls, but mortality is low: NEFOP data from 2002 to 2004 showed 0.2% Atlantic sturgeon mortality in shrimp and otter trawls. The Northern shrimp fishery is not known to interact with ESA-listed cetaceans or sea turtles.

### 5.2.7 American Shad

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

About 40-500 Atlantic sturgeon were reportedly caught in the spring shad gillnet fishery in the past, primarily from the Delaware Bay, with only 2% caught in the river. Effort has more recently switched to striped bass, however. The fishery uses 5-inch mesh gillnets left overnight to soak, but, based on the available information, there is little bycatch mortality of any species in this fishery. Unreported mortality may be occurring in the recreational shad fishery, but the extent is unknown (NMFS 2011a).

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

Several state water recreational shad fisheries (NC, DE, NJ, CT, RI, MA) allow the use of gillnets or pound nets, which have been known to interact with ESA-listed cetaceans and sea turtles, thus interaction may occur where the gear overlaps with sea turtle and cetacean distributions.

All recreational shad fisheries in state waters allow the use of hook and line gear (NC, DE, NJ, CT, RI, MA, NH, ME). 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 (NMFS SEFSC 2001).

#### 5.2.8 Striped Bass

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

Several states have reported incidental catch of Atlantic sturgeon in the striped bass fishery (NMFS 2011a). In southern Maine, the recreational striped bass fishery is known to catch Atlantic sturgeon and in New Hampshire, live bait recreational fisheries are also known to catch Atlantic sturgeon, although numbers are not available. The hook and line striped bass fishery along the south shore of Long Island has recently had reports of sturgeon caught or snagged in recreational gear particularly around Fire Island and Far Rockaway. Atlantic sturgeon bycatch is occurring in the Delaware Bay and River, but little bycatch mortality has been reported. Unreported mortality is likely occurring. In Chesapeake Bay, researchers instituted a reward program for commercial fishermen and received reports of 85 Atlantic sturgeon captured as bycatch in commercial anchored gillnets, primarily in the striped bass fishery, in 2005 and 423 in 2006. Most of the fish came from the James River, followed by the York River, the ocean, and the Rappahannock (Musick and Hager 2007). In North Carolina, the Winter Beach seine fishery for striped bass takes sturgeon (adults and subadults) but has not reported mortalities.

Data from the Atlantic Coast Sturgeon Tagging Database (2000-2004) shows that the striped bass fishery accounted for 43% of Atlantic sturgeon recaptures (ASSRT 2007). The striped bass-weakfish fishery also had one of the highest bycatch rates of 30 directed fisheries according to NEFOP data from 1989 to 2000 (ASSRT 2007). However, greater rates of bycatch do not necessarily translate into high mortality rates. Other factors, such as gear, season, and soak times, may be important variables in understanding Atlantic sturgeon mortality. A recent study on the use of floating gillnets in the striped bass fishery suggests that floating gillnets may reduce bycatch of Atlantic sturgeon while minimally affecting the striped bass catch in Virginia's striped bass fishery (Trice 2011).

State water commercial striped bass fisheries in Delaware, Maryland, Virginia, and North Carolina allow the use of gillnets or trawls, both of which have been known to interact with ESA-listed sea turtles, thus interaction may occur where the gear overlaps with sea turtle distributions. ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear overlaps with cetacean distributions.

All recreational striped bass fisheries in state waters allow the use of hook and line gear (NC, DE, NJ, CT, RI, MA, NH, ME). 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 (NMFS SEFSC 2001).

## 5.3 Impacts of Other Human Activities in the Action Area

### 5.3.1 Maritime Industry

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

### 5.3.2 Pollution

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local or private action, may affect ESA-listed species in the action area. Sources of pollutants in coastal regions of the action area include atmospheric loading of pollutants such as PCBs, storm water runoff from coastal towns, cities and villages, runoff into rivers emptying into bays, groundwater discharges and sewage treatment effluent, and oil spills. The introduction of pollutants, including metals, dioxin, dissolved solids, phenols, and hydrocarbons, from paper mills, sewers, and other industrial sources, may persist in the benthic environment and may affect developing fish eggs and larvae.

Nutrient loading from land-based sources, such as coastal community discharges, is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effect to larger embayments is unknown. Contaminants could indirectly affect ESA-listed species if the pollution reduces the food available to marine animals.

Marine debris (e.g., discarded fishing line, boat lines) can entangle cetaceans or sea turtles causing serious injury or mortality. Turtles commonly ingest plastic or mistake debris for food. Jellyfish are a preferred prey for leatherbacks, and plastic bags, which may look like jellyfish to the turtles, are often found in the turtles' stomach contents (Magnuson *et al.* 1990).

### 5.3.3 Coastal Development

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

### 5.3.4 Catastrophic Events

Commercial vessel traffic/shipping imposes the potential for oil/chemical spills. With human population rising and commerce becoming increasingly globalized, there is more demand for ships. The pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo *et al.* 1986). There have been a number of documented oil spills in the northeastern U.S. Oil spills outside the action area also have the potential to affect ESA-listed species that occur within the action area. For instance, on April 20, 2010 the Deepwater Horizon oil spill occurred in the Gulf of Mexico off the coast of Louisiana. With more than 4.9 million barrels of oil released into an area where ESA-listed species (e.g., loggerhead and Kemp's ridley sea turtles) are known to migrate through, forage, and/or nest along the coastal waters of the Gulf of Mexico, the oil spill is likely to affect their populations; however, because all the information on sea turtle and other ESA-listed species' stranding, deaths, and recoveries has not yet been analyzed, the long-term effects of the oil spill on their populations cannot be determined at this time.

## 5.4 Reducing Threats to ESA-listed Species

### 5.4.1 Education and Outreach Activities

Education and outreach activities are considered some of the primary tools that will effectively reduce the threats to all protected species. For example, NMFS has been active in public outreach to educate fishermen about sea turtle handling and resuscitation techniques, and educates recreational fishermen and boaters on how to avoid 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 strikes to right whales. NMFS also has a program called “SCUTES” (Student Collaborating to Undertake Tracking Efforts for Sturgeon), which offers educational programs and activities about the movements, behaviors, and threats to Atlantic sturgeon. 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.

### 5.4.2 Stranding and Salvage Programs

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

A salvage program is now in place for Atlantic sturgeon. Atlantic sturgeon carcasses can provide pertinent life history data and information on new or evolving threats to Atlantic sturgeon. Their use in scientific research studies can reduce the need to collect live Atlantic sturgeon. The NMFS Sturgeon Salvage Program is a network of individuals qualified to retrieve and/or use Atlantic and shortnose sturgeon carcasses and parts for scientific research and education. All carcasses and parts are retrieved opportunistically and participation in the network is voluntary.

### 5.4.3 Sea Turtle Disentanglement Network (STDN)

NMFS Northeast Region established the Northeast Sea Turtle Disentanglement Network (STDN) in 2002 in response to the high number of leatherback sea turtles found entangled in pot gear along the U.S. Northeast Atlantic coast. The STDN is considered a component of the larger STSSN program, and it operates in all states in the region. The STDN responds to entangled sea turtles and disentangles and releases live animals, thereby reducing serious injury and mortality. In addition, the STDN collects data on live and dead sea turtle entanglement events, providing valuable information for management purposes. The NMFS Northeast Regional Office oversees the STDN program and manages the STDN database.

### 5.4.4 Regulatory Measures for Sea Turtles

#### *5.4.4.1 Large-Mesh Gillnet Requirements in the Mid-Atlantic*

Since 2002, NMFS has regulated the use of large mesh gillnets in federal waters off North Carolina and Virginia (67 FR 13098, March 21, 2002) to reduce the impact of these fisheries on ESA-listed sea turtles. These restrictions were revised in 2006 (73 FR 24776, April 26, 2006). Currently, gillnets with stretched mesh size of 7 inches (17.8 cm) or larger are prohibited in the Exclusive Economic Zone during the following times and in the following areas: (1) north of the NC/SC border to Oregon Inlet, NC 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.

NMFS has also issued regulations to address the interaction 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 inches (10.8 cm) from September 1 through December 15 each year to protect sea turtles. The closed area includes all inshore waters of Pamlico Sound, and all contiguous tidal waters, south of 35E 46.3' N, north of 35E 00' N, and east of 76E 30' W.

#### *5.4.4.2 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 are met (50 CFR 223.206). On February 21, 2003, NMFS issued a final rule to amend the TED regulations to enhance their effectiveness in the Atlantic and Gulf Areas of the southeastern United States by requiring an escape opening designed to exclude leatherbacks as well as large loggerhead and green turtles (68 FR 8456; February 21, 2003). In 2011, NMFS published a Notice of Intent to prepare an Environmental Impact Statement (EIS) and to conduct scoping meetings. NMFS is considering a variety of regulatory measures to reduce the bycatch of threatened and endangered sea turtles in the shrimp fishery of the southeastern United States in light of new concerns regarding the effectiveness of existing TED regulations in protecting sea turtles (76 FR 37050, June 24, 2011).

TEDs are also required for summer flounder trawlers in the summer flounder fishery-sea turtle protection area. This area is bounded on the north by a line extending along 37°05’N (Cape Charles, VA) and on the south by 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).

#### *5.4.4.3 Sea Turtle Conservation Requirements in the Virginia Pound Net Fishery*

NMFS has issued several regulations to help protect sea turtles from entanglement in and impingement on Virginia pound net gear (66 FR 33489, June 22 2001; 67 FR 41196; June 17, 2002; 68 FR 41942, July 16, 2003; 69 FR 24997, May 5, 2004). Currently, all offshore pound leaders in Pound Net Regulated Area I (see Figure 4below) must meet the definition of a modified pound net leader from May 6 through July 15. The modified leader has been found to be effective in reducing sea turtle interactions as compared to the unmodified leader. Nearshore pound net leaders in Pound Net Regulated Area I and all pound net leaders in Pound Net Regulated Area II (see Figure 4 below) must have mesh size less than 12 inches (30.5 centimeters) stretched mesh and may not employ stringers (50 CFR 223.206) from May 6 through July 15 each year. A pound net leader is exempt from these measures only if it meets the definition of a modified pound net leader. In addition, there are monitoring and reporting requirements in this fishery (50 CFR 223.206). As of the 2010 fishing season, 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.

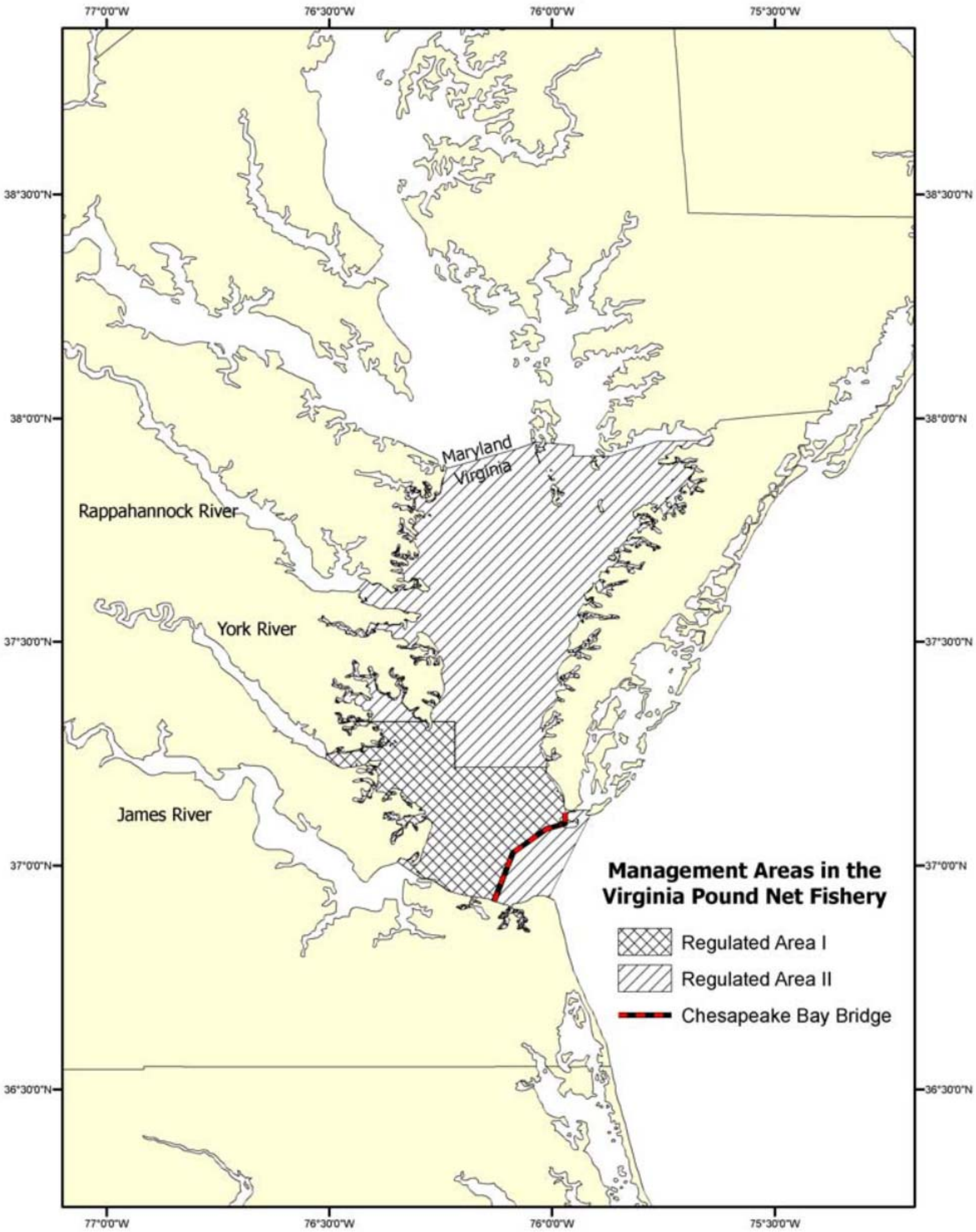


**Figure 4** Management Areas in the Virginia Pound Net Fishery

*5.4.4.4 Sea Turtle Conservation Requirements in the HMS Fishery*

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

In 2008, NMFS SERO completed a section 7 consultation on the continued authorization of HMS Atlantic shark fisheries specifically.To protect declining shark stocks, NMFS sought to greatly reduce the fishing effort in the commercial component of the fishery. These reductions are likely to greatly reduce the interactions between the commercial component of the fishery and sea turtles.

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

##### *5.4.4.5 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, we have 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 (a “chain mat”) over the opening of the dredge bag from of May 1 through November 30 each year. This modification is not expected to reduce the overall number of sea turtle interactions with gear. However, it is expected to reduce the severity of the interactions.

Beginning May 1, 2013, all limited access scallop vessels, as well as Limited Access General Category vessels with a dredge width of 10.5 feet or greater, must use a Turtle Deflector Dredge (TDD) in the Mid-Atlantic (west of 71°W) from May 1 through October 31 each year (77 FR 20728, April 6, 2012). The purpose of the

TDD requirement is to deflect sea turtles over the dredge frame and bag rather than under the cutting bar, so as to reduce sea turtle injuries due to contact with the dredge frame on the ocean bottom (including being crushed under the dredge frame). The TDD has specific components that are defined in the regulations. When combined with the effects of chain mats, which decrease captures in the dredge bag, the TDD should provide greater sea turtle benefits by reducing serious injury and mortality due to interactions with the dredge frame, compared to a standard New Bedford dredge.

##### *5.4.4.6 Sea Turtle Handling and Resuscitation Requirements*

We published as a final rule (66 FR 67495, December 31, 2001) requiring people participating in scientific research or fishing activities to handle and resuscitate (as necessary) incidentally caught 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.

##### *5.4.4.7 Take Exception for Injured, Dead, or Stranded Specimens*

Any agent or employee of NMFS, USFWS, USCG, 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 sea turtle, or dispose of or salvage a dead sea turtle (50 CFR 223.206(b); 50 CFR 222.310). This take exemption extends to our Sea Turtle Stranding and Salvage Network.

### 5.4.5 Atlantic Large Whale Take Reduction Plan

The Atlantic Large Whale Take Reduction Plan (ALWTRP) reduces the risk of serious injury to or mortality of large whales due to incidental entanglement in U.S. commercial trap/pot and gillnet fishing gear. The ALWTRP focuses on the critically endangered North Atlantic right whale, but is also intended to reduce entanglement of endangered humpback and fin whales. The plan is required by the Marine Mammal Protection Act (MMPA) and has been developed by NMFS. The ALWTRP covers the 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.

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. The first ALWTRP went into effect in 1997. For more information on the non-regulatory measures, see the ALWTRP (available online at http://www.nero.noaa.gov/whaletrp/)

*5.4.5.1 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 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 mortalities of right, humpback, and fin whales to insignificant levels approaching zero within five years of its implementation.

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 free itself of the gear, reducing the risk of injury or mortality.
- All groundline must be made of sinking line.

In addition to the regulatory measures implemented to reduce the risk of entanglement in horizontal/groundlines, we, in collaboration with the ALWTRT, have developed a strategy to further reduce risk associated with vertical lines. The actions and timeframe for the implementation of the vertical line strategy is as follows:

- Vertical line model development for all areas to gather as much information as possible regarding the distribution and density of vertical line fishing gear. Status: completed;
- Compile and analyze whale distribution and density data in a manner to overlay with vertical line density data. Status: completed;
- Development of vertical line and whale distribution co-occurrence overlays. Status: completed;
- Develop an ALWTRP monitoring plan designed to track implementation of vertical line strategy, including risk reduction. Status: completed, with annual interim reports beginning in July 2012.
- Analyze and develop potential management measures. Time frame: ongoing;

- Develop and publish proposed rule to implement risk reduction from vertical lines. Time frame: by Mid-2013;
- Develop and publish final rule to implement risk reduction from vertical lines. Time frame: by Mid-2014;
- Implement final rule to implement risk reduction from vertical lines. Time frame: by early 2015.

### 5.4.6 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) 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).

#### *5.4.6.1 Regulatory Measures to Reduce Vessel Strikes to Large Whales*

##### *5.4.6.1.1 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 that had some potential to impede right whale recovery (NMFS 2005a). Following public comment, we 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.

##### *5.4.6.1.2 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 east coast of the U.S.: the right whale feeding grounds in the Northeast and the right whale calving grounds in the Southeast. The USCG worked closely with us 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 and information on precautionary measures to take while in the vicinity of right whales.

###### *5.4.6.1.3 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 a report called "Recommended Measures to Reduce Ship Strikes of North Atlantic Right Whales," which 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 in June 2006 (71 FR 36299; June 26, 2006). We published regulations on October 10, 2008 to implement a 10-knot speed restriction for all vessels 65 feet (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).

SMAs are supplemented by Dynamic Management Areas (DMAs) that are implemented for 15- day periods in areas in which right whales are sighted outside of SMA boundaries. When NOAA aerial surveys or other reliable sources report aggregations of three 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.

The rule will expire five years from the date of effectiveness. NOAA is currently analyzing data on compliance with the rule and the effectiveness of the rule since its implementation to determine the next steps as its expiration in December 2013 approaches.

##### *5.4.6.1.4 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 to reduce vessel strikes. The proposal was submitted to the IMO in April 2006, adopted by the Maritime Safety Committee in December 2006, and 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 further 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 ATBA will be in effect each year from April 1 to July 31, when right whales are known to congregate around the Great South Channel. Implementing this ATBA coupled with narrowing the TSS by one nautical mile will reduce the relative risk of right whale ship strikes by an estimated 74% during April-July (63% from the ATBA and 11% from the narrowing of the TSS).

##### *5.4.6.1.5 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 websites, 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 DMA program, 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.

#### 5.4.7 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 state volunteer stranding networks, biomonitoring, Analytical Quality Assurance for marine mammal tissue samples, a Working Group on Marine Mammal Unusual Mortality Events (UME) and a National Marine Mammal Tissue Bank. Additionally, a serum bank and long-term storage of histopathology tissue are being developed.

#### 5.4.8 Harbor Porpoise Take Reduction Plan (HPTRP)

We have implemented the HPTRP to decrease interactions between harbor porpoise and commercial gillnet gear in waters off New England and the Mid-Atlantic. The HPTRP includes time and area closures and gear modification requirements. Gear modifications include restriction on mesh size, twine size, gillnet floatline length, and requirements to equip gillnets with pingers, among others. Pingers are acoustic deterrent devices. 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. The HPTRP is an evolving plan and amendments have been made as members of the take reduction team, including fishermen, environmental organizations, researchers, and representatives from state and federal government, identify the need for improvements. We published rhe most recent HPTRP amendments in a final rule on February 19, 2010 (75 FR 7383), and included target bycatch rates for different areas and the institution of Consequence Closure Areas if those targets are exceeded.

On October 1, 2012, we announced that the Coastal Gulf of Maine Consequence Closure Area, which spans the coast from Massachusetts to Maine, would be closed to sink gillnets from October 1 through November 30, but then shifted the closure to February 1 through March 31, 2013 (NOAA Northeast Region Bulletin, January 18, 2013) for this year only. This seasonal closure (October-November) will remain in effect until bycatch levels achieve the zero mortality rate goal (ZMRG) established for harbor porpoises or until the HPTRT and NMFS develop and implement new measures. This closure area is being triggered because the average target bycatch rate was exceeded in the first management season by such a margin that, even if the bycatch rate for the second management season was reduced to zero, the average would still exceed the target rate and trigger the closure. The effects of this closure to the fishing industry were evaluated as part of the Environmental Assessment of the modifications to the HPTRP. For more information on the HPTRP including time and area closures visit: www.nero.noaa.gov/hptrp.

#### 5.4.9 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: www.nmfs.noaa.gov/pr/interactions/trt/bdtrp.htm.

#### 5.4.10 Atlantic Trawl Gear Take Reduction Strategy (ATGTRS)

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

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

#### 5.4.11 Magnuson-Stevens Fishery Conservation and Management Act

There are numerous regulations mandated by the Magnuson-Stevens Fishery Conservation and Management Act that may benefit ESA-listed species. Many fisheries are subject to different time and area closures. These area closures can be seasonal or year-round. Closure areas may benefit ESA-listed species due to elimination of active gear in areas where sea turtle and cetaceans are present. However, if closures shift effort to areas with a comparable or higher density of ESA-listed marine mammals,sea turtles, or fish 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: www.nero.noaa.gov/nero/regs/info.html and http://sero.nmfs.noaa.gov/sustainable_fisheries/policy_branch/index.html.

#### 5.4.12 Atlantic Sturgeon Recovery Planning

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

#### 5.4.13 Atlantic Sturgeon International Cooperation

As described in section 4.4, all directed fishing for Atlantic sturgeon has been prohibited in state waters since 1998 and in the U.S. EEZ since 1999. As noted above, a regulated Atlantic sturgeon fishery occurs in Canada and product from the Bay of Fundy fishery, primarily from the Saint John River estuary, is exported (DFO 2011). The CITES requirements for export of Atlantic sturgeon, a CITES Appendix II species, help to ensure the legal origin of marketed Atlantic sturgeon and documentation of any Atlantic sturgeon marketed in the U.S.

#### 5.4.14 Protections for Gulf of Maine Atlantic Sturgeon

The prohibitions listed under section 9(a)(1) of the ESA automatically apply when a species is listed as endangered but not when listed as threatened. When a species is listed as threatened, section 4(d) of the ESA requires the Secretary of Commerce (Secretary) to issue regulations, as deemed necessary and advisable, to provide for the conservation of the species. The Secretary may, with respect to any threatened species, issue regulations that prohibit any act covered under section 9(a)(1). Whether section 9(a)(1) prohibitions are necessary and advisable for a threatened species is largely dependent on the biological status of the species and the potential impacts of various activities on the species. On June 10, 2011, we proposed protective measures for the GOM DPS of Atlantic sturgeon (76 FR 34023).

#### 5.4.15 Research Activity Guidelines

Research activities aid in the conservation of listed species by furthering our understanding of the species' life history and biological requirements. We recognize, however, that many scientific research activities involve capture and may pose some level of risk to individuals or to the species. Therefore, it is necessary for research activities to be carried out in a manner that minimizes the adverse impacts of the activities on individuals and the species while obtaining crucial information that will benefit the species. Guidelines developed by sturgeon researchers in cooperation with NMFS staff (Moser *et al.*, 2000; Damon-Randall *et al.* 2010; Kahn and Mohead 2010) provide standardized research protocols that minimize the risk to sturgeon species from capture, handling and sampling. These guidelines must be followed by any entity receiving a federal permit to do research on Atlantic sturgeon.

#### 5.4.16 Regulatory Actions that Reduce Threats to Atlantic Salmon

We have worked with the Maine Department of Marine Resources (MDMR), FWS, the Penobscot Indian Nation, and other partners to pursue a range of management and research activities to mitigate and reduce the most severe threats to Atlantic salmon and to improve understanding of salmon abundance and population health.

Recovery actions and activities implemented during 2010-2012 included: (1) Conducting reviews of Species Protection Plans for FERC-licensed hydroelectric projects in the GOM DPS; (2) Developing fish passage guidelines; (3) Developing a quantitative model to assess the impacts of proposed dam-related work; (4) Completing a survey of non-power generating dams and their effect on Atlantic salmon habitat that resulted in removal of two dams in 2012, with another four scheduled for 2013; (5) Developing a General Conservation Plan with operating conditions for non-power generating dam owners who request incidental take permits; and (6) Consulting with federal partners to assure that federal actions minimize harm to Atlantic salmon.

#### 5.4.17 International Coordination and Collaboration to Protect Atlantic Salmon

We participate in the North Atlantic Salmon Conservation Organization (NASCO), the international governing body that jointly manages Atlantic salmon. Participation in NASCO has led to the development of multi-year regulatory measures for high-seas Atlantic salmon fisheries, international guidelines for salmon stocking and mitigation of threats from aquaculture practices, and country specific Action Plans that outline the implementation of all the NASCO guidelines.

### 5.4.18 International Atlantic Salmon Research

We work with international partners to conduct annual sampling of the Atlantic salmon fishery in West Greenland. From this sampling, biological information related to the Greenlandic local-use catch is used to confirm catch, support international Atlantic salmon stock assessments, and determine salmon continent-of-origin while providing a platform for research evaluating the ecological health of Atlantic salmon at Greenland.

### 5.4.19 Restoring Ecosystem Function for Atlantic Salmon

NMFS, MDMR, FWS, and other partners have taken a number of steps to restore ecosystem function as part of the Atlantic Salmon Recovery Plan. Among these are dam removals, including the recent removal of the Great Works Dam on the Penobscot River, and the planned removal of the Veazie Dam, the lowermost dam on the Penobscot River. Removal of these two dams allows Atlantic salmon and other diadromous unimpeded access to sections of the Penobscot River that they have not had in 200 years. Several small projects such as bypasses, fishways, culvert replacements, and barrier (including dams) removal helped restore physical and biological features necessary to further salmon recovery in the GOM DPS. In addition, active stocking and fisheries management is supporting recovery of other diadromous species.

### 5.4.20 Atlantic Salmon Annual Assessment and Monitoring

We support several annual assessment and monitoring efforts to gain greater understanding of Atlantic salmon movement patterns and community. This information will help inform future management decisions. Among these efforts are: (1) a satellite-tagging project of adult Atlantic salmon off the coast of West Greenland to track ocean movements; (2) a fish community study in the Penobscot River estuary; and (3) telemetry studies measuring Atlantic salmon smolt survival form the Penobscot River to the Gulf of Maine and monitoring fish at Halifax, Nova Scotia.

## 6.0 Climate Change

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

### 6.1 Background Information on Global Climate Change

The global mean temperature has risen 0.76ºC (1.36°F) over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007a). Precipitation has increased nationally by 5%-10%, mostly due to an increase in heavy downpours (NAST 2000). In comparison, ocean temperatures have only increased by about 0.1ºC (0.18°F) in the last century, with the changes occurring from the surface to depths of about 700 meters (2,300 feet).

There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and other pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b); these trends are most apparent over the past few decades. Information on future impacts of climate change in the action area is discussed below.

Climate model projections exhibit a wide range of plausible scenarios for both temperature and precipitation over the next century. Both of the principal climate models used by the National Assessment Synthesis Team (NAST) project warming in the southeast by the 2090s, but at different rates (NAST 2000): the Canadian model scenario shows the southeast U.S. experiencing a high degree of warming, which translates into lower soil moisture as higher temperatures increase evaporation; the Hadley model scenario projects less warming and a significant increase in precipitation (about 20%). The scenarios examined, which assume no major interventions to reduce continued growth of world greenhouse gases (GHG), indicate that temperatures in the U.S. will rise by about 3$^{o}$-5$^{o}$C (5$^{o}$-9$^{o}$F) on average in the next 100 years, which is more than the projected global increase (NAST 2000). A warming of about 0.2$^{o}$C (0.4°F) per decade is projected for the next two decades over a range of emission scenarios (IPCC 2007). This temperature increase will very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. Climate warming has resulted in increased precipitation, river discharge, and glacial and sea-ice melting (Greene *et al.* 2008).

The past three decades have witnessed major changes in ocean circulation patterns in the Arctic, and these were accompanied by climate associated changes as well (Greene *et al.* 2008). Shifts in atmospheric conditions have altered Arctic Ocean circulation patterns and the export of freshwater to the North Atlantic (Greene *et al.* 2008, IPCC 2006). With respect specifically to the North Atlantic Oscillation (NAO), changes in salinity and temperature are expected to be the result of changes in the earth's atmosphere caused by anthropogenic forces (IPCC 2006). The NAO impacts climate variability throughout the Northern Hemisphere (IPCC 2006). Data from the 1960s through 2006 show that the NAO index increased from minimum values in the 1960s to strongly positive index values in the 1990s, but declined since (IPCC 2006). This warming extends more than 1000 meters (0.62 miles) deep—deeper than anywhere in the world oceans—and is particularly evident under the Gulf Stream/ North Atlantic Current system (IPCC 2006). On a global scale, large discharges of freshwater into the North Atlantic subarctic seas can lead to intense stratification of the upper water column and a disruption of North Atlantic Deepwater (NADW) formation (Greene *et al.* 2008; IPCC 2006). There is evidence that the NADW has already freshened significantly (IPCC 2006). This in turn can lead to a slowing down of the global ocean thermohaline (large-scale circulation in the ocean that transforms low-density upper ocean waters to higher density intermediate and deep waters and returns those waters back to the upper ocean), which can have climatic ramifications for the whole earth system (Greene *et al.* 2008).

While predictions are available regarding potential effects of climate change globally, it is more difficult to assess the potential effects of climate change over the next few decades on smaller geographic scales, such as the Hudson River or Chesapeake Bay, especially as climate variability is a dominant factor in shaping coastal and marine systems. The effects of future change will vary greatly in diverse coastal regions for the U.S. Additional information on potential effects of climate change specific to the action area is discussed below. Warming is very likely to continue in the U.S. over the next 25 to 50 years regardless of reduction in GHG emissions due to emissions that have already occurred (NAST 2000); therefore, it is also expected to continue during the operation of the seven fisheries. It is very likely that the magnitude and frequency of ecosystem changes will increase in the next 25 to 50 years, and it is possible that changes will accelerate. Climate change can cause or exacerbate direct stress on ecosystems through high temperatures, a reduction in water availability, and altered frequency of extreme events and severe storms. Water temperatures in streams and rivers are likely to increase as the climate warms and are very likely to have both direct and indirect effects on aquatic ecosystems. Changes in temperature will be most evident during low flow periods when they are of greatest concern (NAST 2000). In some marine and freshwater systems, shifts in geographic ranges and changes in algal, plankton, and fish abundance are associated with high confidence with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation (IPCC 2007).

A warmer and drier climate is expected to result in reductions in stream flows and increases in water temperatures. Consequences could be a decrease in the amount of DO in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch *et al.* 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer-wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch *et al.* 2000). Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer *et al.* 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change.

While debated, researchers anticipate: 1) the frequency and intensity of droughts and floods will change across the nation; 2) a warming of about 0.2ºC (0.4°F) per decade; and 3) a rise in sea level (NAST 2000). A warmer and drier climate will reduce stream flows and increase water temperature resulting in a decrease of DO and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing. Sea level is expected to continue rising: during the 20th century global sea level has increased 15 to 20 centimeters (6 to 8 inches).

As there is significant uncertainty in the rate and timing of change, as well as the effect of any changes that may be experienced in the action area due to climate change, it is difficult to predict the impact of these changes on whales, sea turtles, Atlantic sturgeon, and Atlantic salmon. The seven fisheries are expected to continue in the near and mid-term future in similar areas, at similar times, and with similar levels of effort, but there is no way to predict at this point in time whether the fishery resources and other environmental conditions will continue to support fisheries that are similar to the proposed action in the long-term future or indefinitely. Since the distribution of effort in the seven fisheries and the status of the resource can change over just a few years, we will primarily consider the effects of climate change on the listed species over the next ten years. Longer-term effects of the fishery and climate change on ESA-listed species, whatever they may be, are speculative and difficult to extrapolate beyond ten years.

## 6.2 Effects of Climate Change on Individual Species

### 6.2.1 Right, Humpback, Sei, and Fin Whales

Whales have persisted for millions of years and throughout this time have experienced wide variations in global climate conditions and have successfully adapted to these changes. Climate change at historical rates (thousands of years) is not thought to have been a problem for whales. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats and potential shifts in the distribution and abundance of prey species. Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (MacLeod 2009). Depending on habitat preferences, changes in water temperature due to climate change may affect the distribution of certain species of cetaceans. For instance, sei, fin, and humpback whales are distributed in all water temperature zones, therefore, it is unlikely that their range will be directly affected by an increase in water temperatures (MacLeod 2009). However, North Atlantic right whales, which currently have a range of sub-polar to sub-tropical, may respond to an increase in water temperature by shifting their range northward, with both the northern and southern limits moving pole-ward.

In regards to marine mammal prey species, there are many potential direct and indirect effects that global climate change may have on prey abundance and distribution, which in turn, poses potential behavioral and physiological effects to marine mammals. For example, 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.

More information is needed in order to determine the potential impacts global climate change will have on the timing and extent of population movements, abundance, recruitment, distribution and species composition of prey (Learmonth *et al.* 2006). Changes in climate patterns, ocean currents, storm frequency, rainfall, salinity, melting ice, and an increase in river inputs/runoff (nutrients and pollutants) will all directly affect the distribution, abundance and migration of prey species (Waluda *et al.* 2001; Tynan and DeMaster 1997; Learmonth *et al.* 2006). These changes will likely have several indirect effects on marine mammals, which may include changes in distribution, including displacement from ideal habitats, decline in fitness of individuals, population size due to the potential loss of foraging opportunities, abundance, migration, community structure, susceptibility to disease and contaminants, and reproductive success (MacLeod 2009). Global climate change may also result in changes to the range and abundance of competitors and predators that will also indirectly affect marine mammals (Learmonth *et al.* 2006).

In the immediate future (2013-2023), it is unlikely that a shift in range will be observed due the extremely small increase, if any, in water temperature predicted to occur in this period. If any shift does occur, it is likely to be minimal and thus, it seems unlikely that this small increase in temperature will cause a significant effect to whales or a significant modification to the number of whales likely to be present in the action area over the life of the proposed action.

### 6.2.2 Sea Turtles

#### *6.2.2.1 Loggerhead Sea Turtles*

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

Climate change has the potential to result in changes at nesting beaches that may affect loggerhead sex ratios. Loggerhead sea turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (*e.g.*, Glen and Mrosovsky 2004; Hawkes *et al.* 2009); however, to the extent that nesting can occur at beaches further north where sand temperatures are not as warm, these effects may be partially offset. The BRT specifically identified climate change as a threat to loggerhead sea turtles in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution. In the threats matrix analysis, climate change was considered for oceanic juveniles and adults and eggs/hatchlings. The report states that for oceanic juveniles and adults, "although the effect of trophic level change from…climate change…is unknown it is believed to be very low." For eggs/hatchlings, the report states that total mortality from anthropogenic causes, including sea level rise resulting from climate change, is considered to be low relative to the entire life stage.

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

In addition, atmospheric warming could cause habitat alteration for food resources such as crabs and other invertebrates. It may increase hurricane activity, leading to an increase in debris in nearshore and offshore waters, which may result in an increase in entanglement, ingestion, or drowning. Increased hurricane activity may cause damage to nesting beaches or inundate nests with sea water. Atmospheric warming may change convergence zones, currents and other oceanographic features that are relevant to loggerhead sea turtles, as well as change rain regimes and levels of nearshore runoff.

#### *6.2.2.2 Kemp's Ridley Sea Turtles*

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

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

Texas coast, and with nesting increasing and the sand temperatures slightly cooler than at Rancho Nuevo, PAIS could become an increasingly important source of males for the population.

Climate change may also affect Kemp's ridleys in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting prey abundance and/or distribution.

In addition, atmospheric warming could cause habitat alteration for food resources such as crabs and other invertebrates. It may increase hurricane activity, leading to an increase in debris in nearshore and offshore waters, which may result in an increase in entanglement, ingestion, or drowning. In addition, increased hurricane activity may cause damage to nesting beaches or inundate nests with sea water. Atmospheric warming may change convergence zones, currents and other oceanographic features that are relevant to loggerhead sea turtles, as well as change rain regimes and levels of nearshore runoff.

#### *6.2.2.3 Leatherback Sea Turtles*

Although leatherbacks are probably already beginning to be affected by impacts associated with anthropogenic climate change in several ways, no significant climate change-related impacts to leatherback turtle populations have been observed to date (PIRO BO 2012). However, over the long term, climate change related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Changes in marine systems associated with rising water temperatures, changes in ice cover, salinity, oxygen levels and circulation including shifts in ranges and changes in algal, plankton, and fish abundance could affect leatherback prey distribution and abundance. Climate change is expected to expand foraging habitats into higher latitude waters and some concern has been noted that increasing temperatures may increase the female to male sex ratio of hatchlings on some beaches (Morosovsky *et al.* 1984 and Hawkes *et al.* 2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b). Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al.* 2009). Leatherbacks have expanded their range in the Atlantic north by 330 kilometers in the last 17 years as warming has caused the northerly migration of the 15°C sea surface temperature (SST) isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary prey, jellyfish, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al.* 1999; Attrill *et al.* 2007;

Richardson *et al.* 2009), which may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited. Even though there may be a benefit to leatherbacks due to climate change influence on productivity, we do not know what impact other climate-related changes may have such as increasing sand temperatures, sea level rise, and increased storm events.

As discussed for loggerheads, increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al.* 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Fish *et al.* 2005). This effect would potentially be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents.

#### *6.2.2.4 Green Sea Turtles*

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

#### *6.2.2.5 Sea Turtle Summary*

As described above, sea turtles are most likely to be affected by climate change due to increasing sand temperatures at nesting beaches, which in turn would result in increased female:male sex ratio among hatchlings; sea level rise, which could result in a reduction or shift in available nesting beach habitat and increased risk of nest inundation; changes in the abundance and distribution of forage species, which could result in changes in the foraging behavior and distribution of sea turtle species; and, changes in water temperature, which could possibly lead to a northward shift in their range.

Over the time period of this action considered in this Opinion, sea surface temperatures are expected to rise less than 1°C. It is unknown if that is enough of a change to contribute to shifts in the range or distribution of sea turtles. Theoretically, we expect that as waters in the action area warm, more sea turtles could be present or sea turtles could be present for longer periods of time. However, if temperature affected the distribution of sea turtle forage in a way that decreased forage in the action area, sea turtles may be less likely to occur in the action area.

It has been speculated that the nesting range of some sea turtle species may shift northward. Nesting in the Mid-Atlantic generally is extremely rare and no nesting has been documented at any beach in the Northeast. In 2010, one green sea turtle came up on the beach in Sea Isle City, NJ; however, it did not lay any eggs. In August 2011, a loggerhead came up on the beach in Stone Harbor, NJ but did not lay any eggs. On August 18, 2011, a green sea turtle laid one nest at Cape Henlopen Beach in Lewes, DE near the entrance to Delaware Bay. The nest contained 190 eggs and was transported indoors to an incubation facility on October 7. A total of 12 eggs hatched, with eight hatchlings surviving. In December, seven of the hatchlings were released in Cape Hatteras, NC. It is important to consider that in order for nesting to be successful in the Mid-Atlantic, fall and winter temperatures need to be warm enough to support the successful rearing of eggs and sea temperatures must be warm enough for hatchlings not to die when they enter the water. Predicted increases in water temperatures between now and 2023 are not great enough to allow successful rearing of sea turtle eggs in the any new parts of the action area. Therefore, it is unlikely that over the time period considered here, that there would be an increase in nesting activity in the action area.

As noted above, sea level rise has the potential to remove possible beach nesting habitat. A recent study by the U.S. Geological Survey found that sea levels in a 620-mile "hot spot" along the East Coast are rising three to four times faster than the global average (Sallenger *et al.* 2012). The disproportionate sea level rise is due to the slowing of Atlantic currents caused by fresh water from the melting of the Greenland Ice Sheet. Sharp rises in sea levels from North Carolina to Massachusetts could threaten wetland and beach habitats, and negatively affect sea turtle nesting along the North Carolina coast. If warming temperatures moved favorable nesting sites northward, it is possible that rises in sea level could constrain the availability of nesting sites on existing beaches. In the next 100 years, the study predicted that sea levels will rise an additional 20-27 centimeters (8-11 inches) along the Atlantic coast "hot spot" (Sallenger *et al.* 2012).

Warming sea temperatures are likely to result in a shift in the seasonal distribution of sea turtles in the action area, such that sea turtles may begin northward migrations from their southern overwintering grounds earlier in the spring and thus would be present in the action area earlier in the year. Likewise, if water temperatures were warmer in the fall, sea turtles could remain in the action area later in the year. In the next ten years, the expected small increase in temperature is unlikely to cause a significant effect to sea turtles or a significant modification to the number of sea turtles likely to be present in the action area.

Changes in water temperature may also alter the forage base and thus, foraging behavior of sea turtles. Changes in the foraging behavior of sea turtles in the action area could lead to either an increase or decrease in the number of sea turtles in the action area, depending on whether there was an increase or decrease in the forage base and/or a seasonal shift in water temperature. For example, if there was a decrease in sea grasses in the action area resulting from increased water temperatures or other climate-change related factors, it is reasonable to expect that there may be a decrease in the number of foraging green sea turtles in the action area. Likewise, if the prey base for loggerhead, Kemp's ridley, or leatherback sea turtles was affected, there may be changes in the abundance and distribution of these species in the action area. However, as noted above, because we do not know the adaptive capacity of these individuals or how much of a change in temperature would be necessary to cause a shift in distribution, it is not possible to predict changes to the foraging behavior of sea turtles over the next ten years. If sea turtle distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact on the availability of food. Similarly, if sea turtles shifted to areas where different forage was available and sea turtles were able to obtain sufficient nutrition from that new source of forage, any effect would be minimal. The greatest potential for effect to forage resources would be if sea turtles shifted to an area or time where insufficient forage was available; however, the likelihood of this happening seems low because sea turtles feed on a wide variety of species and in a wide variety of habitats.

### 6.2.3 Atlantic Sturgeon

Atlantic sturgeon have persisted for millions of years and have experienced wide variations in global climate conditions, to which they have successfully adapted. Climate change at historical rates (thousands of years) is not thought to have been a problem for sturgeon species. However, at the current rate of global climate change, future effects to Atlantic sturgeon are possible. Rising sea level may result in the salt wedge moving upstream in affected rivers. Atlantic sturgeon spawning occurs in fresh water reaches of rivers because early life stages have little to no tolerance for salinity. Similarly, juvenile Atlantic sturgeon have limited tolerance to salinity and remain in waters with little to no salinity. If the salt wedge moves further upstream, Atlantic sturgeon spawning and rearing habitat could be restricted. In river systems with dams or natural falls that are impassable by sturgeon, the extent that spawning or rearing may be shifted upstream to compensate for the shift in the movement of the salt wedge would be limited. While there is an indication that an increase in sea level rise would result in a shift in the location of the salt wedge, at this time there are no predictions on the timing or extent of any shifts that may occur; thus, it is not possible to predict any future loss in spawning or rearing habitat. However, in all river systems, spawning occurs miles upstream of the salt wedge. It is unlikely that shifts in the location of the salt wedge would eliminate freshwater spawning or rearing habitat. If habitat was severely restricted, productivity or survivability may decrease.

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

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

Atlantic sturgeon in the action area are most likely to experience the effects of global climate change in warming water temperatures, which could change their range and migratory patterns. Warming temperatures predicted to occur over the next 100 years would likely result in a northward shift/extension of their range (i.e. into the St. Lawrence River, Canada) while truncating the southern distribution, thus affecting the recruitment and distribution of sturgeon rangewide. In the next ten years, this increase in sea surface temperature is expected to be minimal, and thus, it is unlikely that this expanded range will be observed in the near future. If any shift does occur, it is likely to be minimal and thus, it seems unlikely that this small increase in temperature will cause a significant effect to Atlantic sturgeon or a significant modification to the number of sturgeon likely to be present in the action area over the life of the proposed action. However, even a small increase in temperate can affect DO concentrations. A one degree change in temperature in Chesapeake Bay could make parts of Chesapeake Bay inaccessible to sturgeon due to decreased levels of DO (Batiuk *et al.* 2009).

Although the action area does not include spawning grounds for Atlantic sturgeon, sturgeon are migrating through the action area to reach their natal rivers to spawn. Elevated temperatures could modify cues for spawning migration, resulting in an earlier spawning season, and thus, altering the time of year sturgeon may or may not be present within the action area. This may cause an increase or decrease in the number of sturgeon present in the action area. However, because spawning is not triggered solely by water temperature, but also by day length (which would not be affected by climate change) and river flow (which could be affected by climate change), it is not possible to predict how any change in water temperature alone will affect the seasonal movements of sturgeon through the action area.

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

### 6.2.4 Atlantic Salmon

Atlantic salmon may be especially vulnerable to the effects of climate change in watersheds that are heavily developed and have already been affected by a range of stresses associated with agriculture, industrialization, and urbanization (Elliot *et al.* 1998). Climate effects related to temperature regimes and flow conditions determine juvenile salmon growth and habitat (Friedland 1998). One study conducted in the Connecticut and Penobscot rivers, where temperatures and average discharge rates have been increasing over the last 25 years, found that dates of first capture and median capture dates for Atlantic salmon have shifted earlier by about 0.5 days/ year, and these consistent shifts are correlated with long-term changes in temperature and flow (Juanes *et al.* 2004). This shift in timing illustrates the species adaptability to changing conditions. Temperature increases are also expected to reduce the abundance of salmon returning to home waters, particularly at the southern limits of Atlantic salmon spatial distribution (Beaugrand and Reid 2003).

One recent study conducted in the United Kingdom that used data collected over a 20-year period in the Wye River found Atlantic salmon populations have declined substantially. This decline was best explained by climatic factors, like increasing summer temperatures and reduced discharge, more than any other factor (Clews *et* *al.* 2010). Changes in temperature and flow serve as cues for salmon to migrate, and smolts entering the ocean either too late or too early would then begin their post-smolt year facing less optimal opportunities to feed, predator risks, and/or thermal stress (Friedland 1998). Since the highest mortality affecting Atlantic salmon occurs in the marine phase, both the temperature and the productivity of the coastal environment may be critical to survival (Drinkwater *et al.* 2003). Temperature influences the length of egg incubation periods for salmonids (Elliot *et al.* 1998) and higher water temperatures could accelerate embryo development of salmon and cause premature emergence of fry.

Since fish maintain a body temperature almost identical to their surroundings, thermal changes of a few degrees Celsius can critically affect biological functions in salmonids (NMFS and USFWS 2005). While some fish populations may benefit from an increase in river temperature for greater growth opportunity, there is an optimal temperature range and a limit for growth after which salmonids will stop feeding due to thermal stress (NMFS and USFWS 2005). Thermally stressed salmon also may become more susceptible to mortality from disease (Clews *et al.* 2010). A study performed in New Brunswick found there is much individual variability between Atlantic salmon and their behaviors and noted that the body condition of fish may influence the temperature at which optimal growth and performance occur (Breau *et al.* 2007).

The productivity and feeding conditions in Atlantic salmon's overwintering regions in the ocean are critical in determining the final weight of individual salmon and whether they have sufficient energy to migrate upriver to spawn (Lehodey *et al.* 2006). Survival is inversely related to body size in pelagic fishes, and temperature has a direct effect on growth that will affect growth-related sources of mortality in post-smolts (Friedland 1998). Marine salmon growth increases in a linear trend with temperature, but eventually reaches a maximum rate and decreases at high temperatures (Brett 1979 in Friedland 1998). When at sea, Atlantic salmon eat crustaceans and small fishes, such as herring, sprat, sand-eels, capelin, and small gadids, and when in freshwater, adults do not feed, but juveniles eat aquatic insect larvae (FAO 2012). Species with calcium carbonate skeletons, such as the crustaceans that salmon sometimes eat, are particularly susceptible to ocean acidification, since ocean acidification will reduce the carbonate availability necessary for shell formation (Wood *et al.* 2008). Climate change is likely to affect the abundance, diversity, and composition of plankton, and these changes may have important consequences for higher trophic levels like Atlantic salmon (Beaugrand and Reid 2003).
In addition to temperature, stream flow is also likely to be impacted by climate change and is vital to Atlantic salmon survival.

In-stream flow defines spatial relationships and habitat suitability for Atlantic salmon and since climate is likely to affect in-stream flow, the physiological, behavioral, and feeding-related mechanisms of Atlantic salmon are also likely to be impacted (Friedland 1998). With changes in in-stream flow, salmon found in smaller river systems may experience upstream migrations that are confined to a narrower time frame, as small river systems tend to have lower discharges and more variable flow (Elliot *et al.* 1998). The changes in rainfall patterns expected from climate change and the impact of those rainfall patterns on flows in streams and rivers may severely impact productivity of salmon populations (Friedland 1998). More winter precipitation falling as rain instead of snow can lead to elevated winter peak flows which can scour the streambed and destroy salmon eggs (Battin *et al.* 2007, Elliot *et al.* 1998). Increased sea levels in combination with higher winter river flows could cause degradation of estuarine habitats through increased wave damage during storms (NSTC 2008). Since juvenile Atlantic salmon are known to select stream habitats with particular characteristics, changes in river flow may affect the availability and distribution of preferred habitats (Riley *et al.* 2009). Unfortunately, the critical point at which reductions in flow begin to have a damaging impact on juvenile salmonids is difficult to define, but generally flow levels that promote upstream migration of adults are likely adequate to encourage downstream movement of smolts (Hendry *et al.* 2003).

Humans may also seek to adapt to climate change by manipulating water sources, for example in response to increased irrigation needs, which may further reduce stream flow and biodiversity (Bates *et al.* 2008).Water extraction is a high level threat to Atlantic salmon, as adequate water quantity and quality are critical for all life stages of Atlantic salmon (NMFS and USFWS 2005). Climate change will also affect precipitation, with northern areas predicted to become wetter and southern areas predicted to become drier in the future (Karl *et al.* 2009). Droughts may further exacerbate poor water quality and impede or prevent migration of Atlantic salmon (Riley *et al.* 2009).

It is anticipated that these climate change effects could significantly affect the functioning of the Atlantic salmon critical habitat. Increased temperatures will affect the timing of upstream and downstream migration and make some areas unsuitable as temporary holding and resting areas. Higher temperatures could also reduce the amount of time that conditions are appropriate for migration ($<23^{o}$ Celsius), which could affect an individual's ability to access suitable spawning habitat. In addition, elevated temperatures will make some areas unsuitable for spawning and rearing due to effects to egg and embryo development.

As described above, over the long term, global climate change may affect Atlantic salmon by changing conditions in rivers and oceans. However, there is significant uncertainty, due to a lack of scientific data, on the degree to which these effects may be experienced and the degree to which Atlantic salmon will be able to successfully adapt to any such changes. Any activities occurring within and outside the action area that contribute to global climate change are also expected to affect listed species and their habitat within the action area. While we can make some predictions on the likely effects of climate change on Atlantic salmon, without modeling and additional scientific data, these predictions remain speculative.

Additionally, these predictions do not take into account their adaptive capacity, which determines their ability to deal with change.

## 7.0 Effects of the Proposed Action on ESA-Listed Species

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 Opinion examines the likely effects of the proposed action on ESA-listed species and critical habitat within the action area to determine if the continued operation of the seven fisheries over the next ten years is likely to jeopardize the continued existence of those species or result in the destruction or adverse modification of critical habitat within the next ten years and beyond. This analysis is done after careful review of the status of each listed species and critical habitat and the factors that affect the survival and recovery of those species, as described above. The only critical habitats designated in the action area are for right whales in the North Atlantic, *Acroporid* corals, and Johnson's seagrass. In section 4.0, we determined that the continued operation of the seven fisheries will have no effect on these critical habitats. Therefore, we are only assessing whether the actions under consideration are likely to jeopardize the continued existence of any listed species.

In this section of the Opinion, we will assess the direct and indirect effects of the proposed actions on ESA-listed marine mammals, sea turtles, the five DPSs of Atlantic sturgeon, and the GOM DPS Atlantic salmon that occur in the action area. The purpose of the assessment is to determine if it is reasonable to conclude that the seven fisheries are likely to have direct or indirect effects that appreciably reduce the likelihood of these species surviving and recovering in the wild by reducing their reproduction, numbers, or distribution.

As described in Section 4.0, we have determined that North Atlantic right, humpback, fin, and sei whales; Northwest Atlantic DPS loggerhead, leatherback, Kemp's ridley, and green sea turtles; the GOM, NYB, CB, Carolina, and SA DPSs of Atlantic sturgeon; and GOM DPS Atlantic salmon may be adversely affected by the continued operation of the seven fisheries discussed in this Opinion. Adverse effects will result from interactions with gear used in these fisheries. Our assessment of the effects of interactions with trawl, gillnet, trap/pot, and hook and line gear is provided below.

### 7.1 Approach to the Assessment

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

The final step of the analysis—relating reductions in a species' reproduction, numbers, or distribution to reductions in the species' likelihood of surviving and recovering in the wild—is the most difficult step because (a) the relationship is not linear; (b) to persist over geologic time, most species have evolved to withstand some level of variation in their birth and death rates without a corresponding change in their likelihood of surviving and recovering in the wild; and (c) our knowledge of the population dynamics of other species and their response to human perturbation is usually too limited to support anything more than rough estimates. Nevertheless, our analysis must distinguish between anthropogenic reductions in a species' reproduction, numbers, and distribution that can reasonably be expected to affect the species' likelihood of survival and recovery in the wild and other (natural) declines. 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 have no effect on listed species or critical habitat when, in fact, there would be an effect.

In order to identify, describe, and assess the effects to listed species resulting from interactions with fishing gear used in the seven fisheries considered in this Opinion, we have reviewed information on: (1) entanglements of right, humpback, fin, and sei whales in fishing gear of known and/or unknown origin (Johnson *et al.* 2005; Henry *et al.* 2012; Waring *et al.* 2011; STDN 2012), (2) bycatch of loggerhead, leatherback, Kemp's ridley, and green sea turtles, Atlantic sturgeon, and Atlantic salmon in gillnet and bottom trawl gear in areas where these fisheries operate (Murray 2009a, 2009b; NEFSC 2011; Warden 2011a, 2011b; NEFOP and ASM databases), (3) life history of large whales, sea turtles, Atlantic sturgeon, and Atlantic salmon, and (4) the effects of fishing gear entanglements on large whales, sea turtles, Atlantic sturgeon, and Atlantic salmon that have been published in a number of documents. These sources include status reviews, stock assessments, and biological reports (NMFS and USFWS 1995, 2007a, 2007b, 2007c, 2007d; TEWG 1998, 2000, 2007, 2009; NMFS SEFSC 2001; Moore *et al.* 2004; Stein *et al.* 2004a; Johnson *et al.* 2005; ASMFC TC 2007; ASSRT 2007; Conant *et al.* 2009; Glass *et al.* 2010; Waring *et al.* 2011; Damon-Randall *et al.* 2012a), recovery plans (NMFS 1991a, 1991b, 2005a, 2006, 2011b; NMFS and USFWS 1991, 1992, 2008; NMFS *et al.* 2011), and numerous other sources of information from the published literature as cited within this Opinion.

## 7.2 Interactions between Listed Species and Fishing Gear

### 7.2.1 Factors Affecting Cetacean Interactions by Gear Type

Any line rising into the water column has the potential to entangle a whale (Johnson *et al.* 2005). The general scenario that leads to a whale becoming entangled in gear begins with a whale encountering a line. It may then move along that line until it comes up against something such as a buoy. The buoy can then be caught in the whale's baleen, against a pectoral fin, or on some other body part. When the animal feels the resistance of the gear, it is likely to thrash, which may cause it to become further entangled in the lines associated with gillnet and/or trap/pot gear. For large whales, there are generally three areas of entanglement: (1) the gape of the mouth, (2) around the flippers, and (3) around the tail stock. Right whales spend a substantial amount of time feeding below the surface; this species feeds by swimming continuously with their mouths open. They also roll and lift their flippers about the water's surface, behaviors that may add to entanglement risk, especially from vertical buoy lines and surface system lines. Humpback whales commonly use their mouths, flippers, and tails to aid in feeding. Thus, while foraging, all body parts are at risk of entanglement.

Susceptibility to entanglement depends on a species' physical characteristics and behavior. The probability that a large whale will initially survive an entanglement in fishing gear depends on the species and age of the individual involved. This is due in part to variations in size, diving behavior, and foraging behavior, as well as to location and time of the entanglement. If the gear attached to the line is too heavy for the whale, drowning may result. However, many whales have been observed swimming with portions of line, with or without additional fishing gear, wrapped around a pectoral fin, the tail stock, the neck, or the mouth. Documented cases show that entangled whales may travel for extended periods of time and over long distances before freeing themselves, being disentangled by humans, or dying as a result of the entanglement (Angliss and Demaster 1998). Entanglement may lead to exhaustion and starvation due to increased drag (Wallace 1985). Other effects include infections and deformations. A sustained stress response, such as repeated or prolonged entanglement in gear, makes large whales less able to fight infection or disease, and may make them more prone to ship strikes. Younger animals are particularly at risk if the entangling gear is tightly wrapped since the gear will become more constricting as they grow. The majority of large whales that become entangled are juveniles (Angliss and Demaster 1998). Factors affecting large whale interactions with fishing gear from the seven fisheries are: (1) overlap of whales in time and space with the seven fisheries and, (2) type of gear.

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). The highest abundances of North Atlantic right, humpback, fin, and sei whale populations occur from March through November in New England waters, which is also the peak fishing period for gillnet and bottom trawl gear for the seven fisheries in these waters. Humpback and fin whales are present in Mid-Atlantic waters from October-March in seemingly increasing numbers. Low numbers of whales are present in New England waters through the winter with a portion of the right whale population appearing to remain in the Gulf of Maine over winter (NEFSC unpublished data). Because of substantial interannual and geographic variation in whale occurrences and lack of complete data for seasonal distributions, we consider the potential for whale interactions with the seven fisheries 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 seven fisheries operate, North Atlantic right, humpback, fin, and sei whales are most likely to overlap with operation of the seven fisheries from May through November in New England waters and throughout the fall and winter in Mid-Atlantic and as far south as northern Florida waters. Bottom otter trawl use is highest in the spring months, though is still in use throughout the summer and fall. Gillnet use peaks in the summer for the seven fisheries (NEFMC 2000).

It is often difficult to assign gear found on stranded animals or observed on animals at sea to a specific fishery. Only a fraction of the interactions between large whales and fishing gear are reported. Consequently, documented interactions are an underestimate of the total level of interactions, which cannot be determined through extrapolation.

Due to their size, right, humpback, fin, and sei whales are extremely unlikely to be captured in bottom otter trawl gear. As stated in Section 6.1.2, there have been no documented interactions between right, humpback, fin, and sei whales and the North Atlantic bottom trawl fishery (NEFSC FSB 2011). Their great size and mobility presumably allows them to avoid interactions with the relatively slow moving trawl gear.

Large whales are vulnerable to entanglement in vertical or ground lines associated with sink gillnet and trap/pot gear. Interactions between these species and gillnet/trap/pot gear used in these fisheries 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 hinder the ability of the line to pass through the baleen. 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, yet the extra resistance could conversely increase the effectiveness of weak links to assist in shedding gear from entangled whales.

#### 7.2.2 Description of Existing Information on Interactions with Cetaceans

The NMFS manages the most complete and up to date large whale entanglement data set, which includes data from the Atlantic Large Whale Disentanglement Network (ALWDN), NEFOP, and ASM. The ALWDN receives reports from a variety of sources, such as recreational boaters, commercial fishermen, USCG, NMFS aerial surveys, and research vessels. The MMHSRP also contributes to the collection of fishery interaction data. The Marine Mammal Stranding Network evaluates stranded cetaceans and determines if commercial fishing activity was involved. NMFS has collectively analyzed both datasets and a summary is presented below.

The table below summarizes documented fishing gear interactions with large whales in the Atlantic for 2006-2010, showing the number of documented entanglements, and how many of those led to serious injury or mortality (NMFS NERO 2012).

**Table 24** NMFS gear analysis for entangled/entrapped North Atlantic right, humpback, fin, and sei whales for the years 2006-2010. (For the purposes of this evaluation, entanglement/entrapment events with gear determined to be from Canadian fisheries were not included. The criteria used to categorize these events to U.S., Canada, or undefined origin were results of gear analyses; where not known, the NOAA Stock Assessment Reports for Marine Mammals use the location the animal was first sighted, which may be quite a distance from the original location of entanglement. For this analysis, animals entangled in gear of undefined origin are assumed to be entangled in gear from U.S. fisheries. Confirmed serious injury/mortality (SI/M) events are presented in parentheses.)

| | Entanglement events with gear of U.S. and unidentified origins | # of North Atlantic right whale events | Mean annual North Atlantic right whale events | # of humpback whale events | Mean annual humpback whale events | # of fin whale events | Mean annual fin whale events | # of sei whale events | Mean annual sei whale events |
|---|---|---|---|---|---|---|---|---|---|
| Sink gillnet gear | 5 | 0 | 0 | 5(1) | 1(0.2) | 0 | 0 | 0 | 0 |
| Unspecified gillnet gear | 2(1) | 1(1) | 0.2(0.2) | 1 | 0.2 | 0 | 0 | 0 | 0 |
| Lobster gear | 10(2) | 1 | 0.2 | 9(2) | 1.8(0.4) | 0 | 0 | 0 | 0 |
| Other pot/trap gear | 2(2) | 2(2) | 0.4(0.4) | 0 | 0 | 0 | 0 | 0 | 0 |
| Hook and line | 7 | 0 | 0 | 7 | 1.4 | 0 | 0 | 0 | 0 |
| Bottom longline | 1 | 1 | 0.2 | 0 | 0 | 0 | 0 | 0 | 0 |
| Purse seine | 1 | 0 | 0 | 1 | 0.2 | 0 | 0 | 0 | 0 |
| Unknown gear | 101(30) | 23(4) | 4.6(0.8) | 63 (19) | 12.6(3.8) | 12(4) | 2.4(0.8) | 3(3) | 0.6(0.6) |
| Totals | 129(34) | 28(6) | 5.6(1.2) | 86(22) | 17.2(4.4) | 12(4) | 2.4(0.8) | 3(3) | 0.6(0.6) |

To look at the range of entanglements that may result in SI/M per year as a result of U.S. fishing gear, we looked at the past 10 years of data to increase the sample size.

Between 2001 and 2010, the annual ranges of entanglements resulting in SI/M as a result of U.S. fishing gear was zero to three for North Atlantic right whales, zero to three for fin whales, zero to eight for humpback whales, and zero to two for sei whales (NMFS NERO 2012).

Serious injury has been defined in 50 CFR 229.2 as an injury that is likely to lead to mortality. Currently, NMFS Regional Offices and Science Centers use regional techniques for assessing and quantifying the serious injuries of marine mammals based on the results of a 1997 workshop (Angliss and DeMaster 1998). Although these regional techniques help to accomplish the MMPA's mandates, NMFS recognizes the need for a nationally consistent and transparent process of determining SI for effective conservation of marine mammal stocks and management of human activities impacting these stocks. NMFS convened a Serious Injury Technical Workshop in 2007 to review performance under existing processes, and gather the best available and current scientific information (Andersen *et al.*, 2008).

Based on results of the 2007 workshop and input from marine mammal scientists, veterinary experts, and the MMPA Scientific Review Groups, NMFS has developed a policy and procedural directives describing national guidance and criteria for distinguishing serious from non-serious injuries of marine mammals (76 FR 42116, July 18, 2011). The directives serve as the basis for analyzing marine mammal injury reports (*e.g.*, observer, disentanglement, and stranding program reports) and incorporating the results into marine mammal stock assessment reports (SAR) and marine mammal conservation management regimes (*e.g.*, MMPA List of Fisheries (LOF), take reduction plans (TRP), ship speed regulations). The directives will ensure the consistent interpretation of what constitutes a serious injury and addresses the issues of accounting for injury cases where the outcome cannot be determined as well as accounting for successful mitigation efforts. The national standard federal register notice was published on January 23, 2012 (77 FR 3233, January 23, 2012). Historic serious injury information is expected to change the NEFSC SI/M data in the future. However the historic SI/M information has not yet been changed. Therefore, in this Opinion, current NEFSC SI/M data will be used.

There have been seven documented interactions between humpback whales and hook and line gear; none of these were documented as serious injuries or mortalities. Interactions with hook and line gear and right, fin, and sei whales have not been reported.

From January 2006 through December 2010, there were four verified entanglements of humpback whales in sink gillnet gear that was assessed to be U.S. origin and one entanglement in this gear where the country of origin was not definable (NMFS NERO 2012). One of these interactions resulted in serious injury (NMFS NERO 2012). Within the same time period, two additional entanglements (one humpback, one right whale) were documented with gillnets without specific classification of the type of gillnet.

From January 2006 through December 2010, there were verified entanglements of one right whale and nine humpback whales in U.S. origin lobster gear or has not been identified to a country of origin (NMFS NERO 2012). Although the lobster fishery is not included in this Opinion, some of the seven fisheries are known to use gear similar to that used in the lobster fishery (*i.e.,* trap/pot).

Since many entanglement events go unobserved and because the gear type, fishery, and/or country of origin for reported entanglement events are often not traceable; the list of identified entanglement events is assumed to be an under-representation of actual numbers of entanglements.

There is information that needs to be considered when SI/M and identified gear are looked at together. The identified gear is only looking at gear recovered or identified in the field by markings from the entanglement case. Frequently, entangled whales have numerous physical body locations of entanglement trauma without gear present; this means that the original entanglement configuration is no longer present and has changed since the first observation. Portions of the gear such as weak links and even the physical struggle of the initial entanglement could break free portions of the gear. For example, if an entanglement case had recovered sinking groundline, it is possible that the animal could have become entangled in other parts of the gear and carried off a significant portion of the entire set, with the sinking line being the only part recovered. Also, although uncommon, gear is sometimes lost during disentanglement operations.

Large whale data for 2011 and 2012 are presented below. These data are preliminary and often change before they are finalized when cases are looked at more thoroughly, therefore, these data will not be considered in this Opinion. We expect that these data will not be finalized until after the consultation is complete; expected changes to this data include the addition or deletion of cases and altering the determination or status of any case. Cases include animals that had gear present. Deceased animals that had entanglement trauma but no gear present are not included in these numbers. Reported numbers should be considered a minimum number and not comprehensive.

## 2011 Preliminary Large Whale Data

**Table 25** 2011 Preliminary Large Whale Entanglement Summary[1]
United States and Canadian Waters

| | **Reports of Individual Animals with Previously Unreported Entanglements**[2] |
|---|---|
| Right Whale | 11 |
| Humpback Whale | 19 |
| Fin Whale | 1 |
| Sei Whale | 0 |
| Minke Whale | 5 |
| Sperm Whale | 0 |
| Unknown Beaked Whale | 0 |
| Bryde's Whale | 0 |
| Unknown | 0 |
| TOTAL | 36 |

[1] This is preliminary data and has not been formally disseminated by the National Marine Fisheries Service. It does not represent and should not be construed to represent any agency determination or policy. Additional information gathered after the release of this summary may alter, add or delete cases.
[2] Numbers include live and dead animals

There were 36 whales that were reported and confirmed entangled by survey aircraft, fishermen, whale watch vessels and various other sources within the United States and Canadian waters in 2011. The reports of animals within Canadian waters should not be considered comprehensive due to uncertainty. Of the 36 individuals, 12 of the animals were assessed and responded to; the remaining animals were not responded to due to the fact that they were lost by the reporting platform, were not found by the responder (typically because no one stood by), conditions (sea state, time of day, range offshore) did not allow a response, animal was deceased or were reported to have a minor entanglement or shed the gear during the initial observation of the animal.

Breakdowns of the first and important sightings of new entangled cases are listed below (identification of individual is unknown unless stated):

- **Humpback whale on 01/07/11**
- **Right whale #3010 (mother) on 01/19/11, gear shed**
- **Right whale #3712 on 01/30/11, gear shed**

- **Humpback whale “EKG” on 02/01/11**
- **Right whale #3760 on 02/13/11, gear shed**
- Right whale #3993 on 02/13/11
- Right whale on 3/16/11, deceased
- **Right whale #3893 on 3/17/11, gear shed**
- Humpback whale on 4/11/11
- Humpback whale on 4/15/11, gear shed
- **Right whale #4040 on 4/22/11, disentangled**
- Right whale #3302 on 4/22/11
- Right whale #3123 on 4/29/11, gear shed
- Minke whale on 5/6/11, deceased
- Humpback whale on 5/30/11
- **Humpback whale, 2009 calf of “Lavalier” on 6/3/11, disentangled**
- Humpback whale on 7/9/11
- Finback whale on 7/9/11, gear shed
- Humpback whale on 7/10/11
- Minke whale on 7/17/11
- **Humpback whale “Reflection” on 7/18/11, disentangled**
- Humpback whale on 7/21/11
- Minke whale on 7/24/11
- **Humpback whale “Ganesh” on 7/25/11, gear shed**
- Humpback whale “Reflection” (new entanglement) on 7/30/11, gear shed
- Humpback whale, 2009 calf of “Rapier” on 7/30/11, gear shed
- Humpback whale, 2011 calf of “Canopy” on 7/31/11, gear shed
- Humpback whale “Artillery” on 8/2/11, gear shed
- Humpback whale “Echo” on 8/14/11
- Humpback whale “Checkmark” gear shed
- Right whale, 2010 calf of #3360, on 9/18/11
- Right whale #3111 on 9/27/11
- **Humpback whale “Hippocampus” on 9/30/11, disentangled**
- **Minke whale on 10/5/11, disentangled**
- Minke whale on 10/6/11, deceased
- Humpback whale “Clutter” on 10/10/11

* Cases in bold are when a disentanglement response was possible. Some gear may have been removed in previous sightings which could have lead to a gear free status or the whale with some entangling gear remaining. Gear remaining on a whale does not necessarily mean the whale is in a life-threatening entanglement.

## 2012 Preliminary Large Whale Data

**Table 26** 2012* Preliminary Large Whale Entanglement Summary[1] United States and Canadian Waters

| | Reports of Individual Animals with Previously Unreported Entanglements[2] |
|---|---|
| Right Whale | 5 |
| Humpback Whale | 23 |
| Finback Whale | 2 |
| Sei Whale | 0 |
| Minke Whale | 7 |
| Sperm Whale | 0 |
| Unknown Beaked Whale | 0 |
| Bryde's Whale | 0 |
| Unknown | 0 |
| TOTAL | 37 |
| * Up to and including September 21, 2012 | |

[1] This is preliminary data and has not been formally disseminated by the National Marine Fisheries Service. It does not represent and should not be construed to represent any agency determination or policy. Additional information gathered after the release of this summary may alter, add or delete cases.
[2] Numbers include live and dead animals

As of September 21, 2012, there were 37 whales that were reported and confirmed entangled by survey aircraft, fishermen, whale watch vessels and various other sources within the United States and Canadian waters in 2012. The reports of animals within Canadian waters should not be considered comprehensive due to uncertainty. Of the 37 individuals, 18 of the animals were assessed and responded to; the remaining animals were not responded to due to the fact that they were lost by the reporting platform, were not found by the responder (typically because no one stood by), conditions (sea state, time of day, range offshore) did not allow a response, animal was deceased or were reported to have a minor entanglement or shed the gear during the initial observation of the animal.

Breakdowns of the first and important sightings of new entangled cases are listed below (identification of individual is unknown unless stated):

- Right whale #3821 on 01/07/12
- Right whale # 1719 on 1/19/12
- Humpback whale on 1/26/12

- Minke whale on 2/4/12
- **Right whale #3996 on 2/15/12**
- **Humpback whale on 3/11/12, partially disentangled**
- **Humpback whale on 4/7/12, disentangled**
- **Humpback whale (same animal as 4/7/12; new case) on 4/13/12, partially disentangled**
- Humpback whale on 4/29/12
- Right whale on 5/15/12
- **Humpback whale “Basmati” on 5/17/12, gear shed**
- Humpback whale “Etch-A-Sketch” on 6/9/12, gear shed
- Humpback whale “Apex” on 6/9/12
- **Humpback whale “Sabot” on 6/18/12, partially disentangled**
- **Minke whale on 6/21/12**
- **Humpback whale “Dome” on 6/22/12, gear shed**
- Minke whale on 7/1/12
- **Humpback whale “Hiatus” on 7/5/12, disentangled**
- **Humpback whale “Serengeti” on 7/6/12, disentangled**
- **Humpback whale “Piano” on 7/8/12, gear shed**
- **Minke whale on 7/13/12, partially disentangled**
- Finback whale on 7/16/12
- Minke whale on 7/17/12
- Right whale on 7/19/12, deceased
- **Humpback whale on 7/29/12**
- Finback whale (Blue Ocean Society #0631) on 7/30/12
- Minke whale on 8/2/12
- Humpback whale “Aphid” on 8/4/12
- Humpback whale “Doric” on 8/17/12
- **Humpback whale “Hiatus” on 8/18/12**
- Humpback whale (2011 calf of “Wizard”) on 8/21/12
- **Minke whale on 8/22/12, disentangled**
- **Humpback whale “Forceps” on 8/242/12**
- Humpback whale “Cardhu” on 8/27/12
- Humpback whale “Reflection” on 9/3/12, gear shed
- **Humpback whale on 9/16/12, disentangled**
- **Humpback whale on 9/17/12, partially disentangled**

* Cases in bold are when a disentanglement response was possible. Some gear may have been removed in previous sightings which could have led to a gear free status or the whale with some entangling gear remaining. Gear remaining on a whale does not necessarily mean the whale is in a life-threatening entanglement.

Because whales often free themselves of gear following an entanglement event, scarring may be another useful indicator in monitoring fisheries interactions with large whales. A study conducted by Robbins et al. (2009) analyzed entanglement scars observed in photographs taken during 2003-2006. This analysis suggests high rates of entanglements of Gulf of Maine humpback whales in fishing gear. In an analysis of the scarification of right whales, 358 of 493 (72.6%) whales examined during 1980-2004 were scarred at least once by fishing gear (Knowlton *et al.* 2008). On November 9, 2009, NMFS convened a workshop of the Atlantic Large Whale Take Reduction Team Scarring Rates Work Group to examine the potential of utilizing scarring rates as an ALWTRP monitoring metric. Workshop conclusions recommended continued research on analyzing scarring rates for use in ALWTRP monitoring. NMFS continues to support and monitor research on methods to determine how analyses of scarring rates can best support conservation objectives, as outlined in the ALWTRP Monitoring Strategy that has been developed by NMFS. However, at this time we are not able to use scarification data to determine the number of past entanglements or to predict the likely rate of entanglements in the future.

As noted previously, reported entanglement events are not a complete count of all entanglements that occur. We do not currently have an accepted method to extrapolate those reported events to obtain a complete count estimate. For that reason, the reported 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. For the period 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 total number 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 of the total number of entanglements. We also note that the estimate using this scarification approach indicates that the number of entanglements may be significantly higher than is reported and provides further support for ongoing efforts to implement and enhance risk reduction measures.

### 7.2.3 Factors Affecting Sea Turtle Interactions by Gear Type

The primary factors affecting sea turtle interactions with the seven fisheries are (1) overlap in time and space, (2) method of fishing, (3) the behavior of sea turtles in the presence of gear, and (4) oceanographic features.

As described in the **Status of the Species**, the occurrence of loggerhead, leatherback, Kemp's ridley, and green sea turtles in Northwest Atlantic waters is primarily temperature dependent (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005a; Braun-McNeill *et al.* 2008). In general, sea turtles move up the U.S. Atlantic coast from southern wintering areas as water temperatures warm in the spring (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005a; Braun-McNeill *et al.* 2008). The trend is reversed in the fall as water temperatures cool. In the vicinity of Cape Hatteras during late fall and early winter, the narrowness of the continental shelf and influence of the Gulf Stream helps to concentrate sea turtles, making them more susceptible to fishery interactions (Epperly *et al.* 1995a). By December, sea turtles have passed Cape Hatteras, returning to warmer waters of the U.S. South Atlantic and Gulf of Mexico for the winter (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005a). Recreational anglers have reported sightings of sea turtles in inshore waters (bays, inlets, rivers, or sounds) as far north as New York as early as March-April, but in relatively low numbers (Braun-McNeill and Epperly 2004). Greater numbers of loggerheads, Kemp's ridleys, and greens are found in inshore, nearshore, and offshore waters of the southern Mid-Atlantic (Virginia and North Carolina) from May-November (Mansfield *et al.* 2009) and in inshore, nearshore, and offshore waters of the northern Mid-Atlantic (New York and New Jersey) from June-October (Keinath *et al.* 1987; Morreale and Standora 1993; Braun-McNeill and Epperly 2004). Hard-shelled sea turtles are more commonly found in waters south of Cape Cod, but may also occur in waters farther north (Morreale and Standora 1998). Leatherback sea turtles have a similar seasonal distribution, but have a more extensive range into the Gulf of Maine compared to the hard-shelled sea turtle species (Shoop and Kenney 1992; Mitchell *et al.* 2003; STSSN database).

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

In the summer of 2010, as part of the AMAPPS project, the NEFSC and SEFSC estimated the abundance of juvenile and adult loggerhead sea turtles in the portion of the northwestern Atlantic continental shelf between Cape Canaveral, Florida and the mouth of the Gulf of St. Lawrence, Canada. The abundance estimates were based on data collected from an aerial line-transect sighting survey as well as satellite tagged loggerheads. The preliminary regional abundance estimate was about 588,000 individuals (approximate inter-quartile range of 382,000-817,000) based on only the positively identified loggerhead sightings, and about 801,000 individuals (approximate inter-quartile range of 521,000-1,111,000) when based on the positively identified loggerheads and a portion of the unidentified sea turtle sightings (NEFSC 2011a). The satellite tracks of loggerheads studied as part of the AMAPPS program can be found at http://www.seaturtle.org/tracking/?project_id=537&dyn=1324309895.

Sea turtle interactions with gillnet and trawl gear used in these fisheries 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 can 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 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 serious risk to sea turtles, constriction of a sea turtle's neck and flippers can lead to infection or amputation of limbs, which may result in mortality or impaired foraging or swimming ability. Sea turtles that do escape often retain pieces of gear that can inhibit their foraging or survival. If the turtle is released or escapes with line attached, the flipper may eventually become occluded, infected, and necrotic. Size of the gear (*e.g.*, mesh size), duration of sets/tows, and effectiveness of gear modifications (TEDs in trawls) will influence the likelihood of serious injury and mortality to sea turtles that are incidentally caught (Epperly et al. 2002, need other references).

Sea turtles (primarily leatherbacks, greens, and loggerheads, based on available entanglement data) are also vulnerable to entanglement in pot/trap gear. Leatherbacks may be attracted to the buoys which could appear as jellyfish, or they may be attracted to the organisms which colonize ropes and buoys and could serve as prey. While it is unlikely that loggerheads are attracted to the buoys as prey, loggerheads have been known to become entangled in pot/trap gear as well. Records of stranded or entangled sea turtles indicate entanglement of pot/trap lines around the neck, flipper, or body of the sea turtle; these entanglements can severely restrict swimming or feeding (Balazs 1985). Drowning may occur quickly if the weight of the gear prevents the turtle from reaching the surface to breath or, at a later time, if trailing gear becomes lodged between rocks and ledges below the surface. Leatherbacks may be more susceptible to drowning as compared to other sea turtles due to their unusual physiology and metabolic processes (Lutcavage and Lutz 1997). Leatherbacks lack calcium, which aids in the neutralizing of lactic acid that builds up by increasing bicarbonate levels. The dive behavior of leatherbacks consists of continuous aerobic activity. When entanglement occurs, available oxygen decreases allowing anaerobic glycolysis to take over producing high levels of lactic acid in the blood (Lutcavage and Lutz 1997). Therefore, especially when caught, the stored oxygen is likely to be used up quickly. Anecdotal evidence indicates that when leatherbacks encounter trap/pot gear, they may swim in circles resulting in multiple wraps around a flipper. Long pectoral flippers along with extremely active behavior may make leatherback sea turtles especially vulnerable to entanglement. The softer epidermal tissue of leatherbacks may also make them more susceptible to serious injuries from entangling gear. As with gillnet gear, constriction of a turtle’s neck or flippers can lead to serious injury or mortality. While drowning is the most serious consequence of entanglement, 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. If the turtle escapes or is released from the gear with line attached, the flipper may eventually become occluded, infected, and necrotic. Entangled sea turtles can also be more vulnerable to collision with boats, particularly if the entanglement occurs at or near the surface (Lutcavage *et al.* 1997).

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. Loggerheads and Kemp's ridleys are the species caught most often; these turtles frequently ingest the hooks. Hooked sea 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 STSSN, have occurred during pier fishing. Deceased sea turtles found stranded with hooks in their digestive tract have been reported, although 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 (which may lead to limb amputations) or around seafloor obstructions, preventing the animal from surfacing (leading to drowning). Thus, some of these hooking/entanglement interactions may eventually be lethal.

In regards to the recreational component of the seven 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. Therefore, NMFS is unable to estimate an amount or extent of take occurring in the recreational component of the seven fisheries at this time and will instead focus the majority of the effects analysis on the commercial component of the fishery. In order to better understand the impacts of recreational fishing on sea turtles, NMFS has initiated a survey-based pilotstudy, which was initiated in 2012, and will be ongoing through 2013. This pilot study will assess the extent of interactions between recreational anglers and sea turtles, and includes shore-based, private vessel, and charter/headboat fishing effort. The pilot study for this work has been conducted in the southeast Atlantic states.

Documented cases have indicated that entangled 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.

The behavior of sea turtles in the presence of fishing gear also affects interactions. Video footage recorded by the SEFSC’s Pascagoula Laboratory showed that loggerhead sea turtles will keep swimming in front of an advancing shrimp trawl, rather than deviating to the side, until the turtles become fatigued and are caught by the trawl or the trawl is hauled up (NMFS 2002a). However, it was later determined that the available data were inconclusive and that sometimes sea turtles remained on the bottom, while others shot to the top with bottom disturbance from trawl gear (J. Mitchell pers. comm. in DeAlteris 2010). There was also additional discussion about whether sea turtle behavior in front of approaching trawl gear was more indicative of how long it had been since the turtle had last surfaced for air.

Starting in 2007, Coonamessett Farm began a series of research projects to assess and implement the use of a remotely operated vehicle (ROV) to observe sea turtle behavior in the water column and on the sea floor in the Mid-Atlantic. The ROV studies focused on Atlantic sea scallop fishing grounds with water depths of 40-80 meters during the months of June (2008, 2009), July (2009), August, (2008) and September (2007, 2009) (Smolowitz and Weeks 2009, 2010; Weeks *et al.* 2010). During these studies, over 50 sea turtles were tracked by ROV for periods ranging from two minutes to over eight hours (Smolowitz and Weeks 2009; Weeks *et al.* 2010). In addition to footage collected from the ROV, visual observations and recordings from the masthead were obtained. A range of loggerhead behaviors were observed, including feeding, diving, swimming, surface, and social behaviors. Loggerheads were observed feeding on jellyfish within the top ten meters of the surface and on crabs and scallops on the ocean bottom (Smolowitz and Weeks 2009; Weeks *et al.* 2010). A number of sea turtles were recorded on the ocean bottom at depths of 49-70 meters, and water temperatures of 7.5°-11.5°C (Smolowitz and Weeks 2009, 2010; Weeks *et al.* 2010). Bottom times in excess of 30 minutes were recorded (Weeks *et al.* 2010).

With respect to oceanographic features, a review of the data associated with 11 sea turtles captured by the scallop dredge fishery in 2001 concluded that the captured sea turtles appeared to have been near the shelf/slope front (D. Mountain, pers. comm.). Intensity of biological activity in the Northwest Atlantic 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 cooler, fresher waters from areas to the north (*e.g.*, the Scotian Shelf; Townsend *et al.* 2006). There may be an increased risk of interactions between sea turtles and fishing gear in areas where these oceanographic features occur simply because there are more sea turtles and possibly more fishing gear present which increases the potential for interactions. However, at present we are unable to determine if any of these oceanographic features affect the likelihood of interactions between sea turtles and the seven fisheries. As discussed later on in this section, variables such as latitude, bottom depth, and sea surface temperature have been correlated with sea turtle interaction rates with gillnet and bottom trawl gear in the Mid-Atlantic (Murray 2009b; Warden 2011b).

Given the seasonal distribution of sea turtles and the times and areas when the seven fisheries operate, all four species of sea turtles are likely to overlap with operation of the seven fisheries primarily from May through November in U.S. Mid- and South Atlantic waters as well as around Georges Bank and in the Gulf of Maine. Loggerhead interactions are possible year-round in the southern portion of the Mid-Atlantic (Murray and Orphanides 2013). Interactions with other sea turtle species outside these months and in other portions of the action area are certainly possible, albeit at lower frequencies.

#### 7.2.4 Description of Existing Information on Interactions with Sea Turtles

The discussion of sea turtle interactions that follows will focus on gillnet, trawl, trap/pot, and hook and line (longline and handline) gear. Sea turtles incidentally captured or entangled in these types of fishing gear must be reported to NMFS on VTRs that are required for most Federal fisheries with the exception of the American lobster fishery. At present, compliance with the requirement for federally permitted fishermen to report sea turtle interactions on their VTRs is believed to be very low (as evidenced by the lack of reported interactions that have been documented on vessels with observers in recent years). Without reliable VTR reporting of sea turtle interactions, we are using information on gillnet, bottom trawl, and hook and line interactions collected through the NEFOP and ASM programs, managed through the NEFSC FSB. Both of these programs collect, process, and manage data and biological samples obtained by trained observers during commercial fishing trips throughout the New England and the Mid-Atlantic regions. For trap/pot gear interactions, we also reviewed sea turtle entanglement data that has been collected through and provided by Northeast Region STDN because the NEFOP and ASM programs observe very few trap/pot trips.

Past observed interactions of sea turtles in these three gear types were reviewed in the individual 2010 Opinions for these seven fisheries. Updated information is provided herein. The number of reported interactions is a fraction of the total amount occurring, which is largely unknown for most species. However, in the case of loggerhead sea turtles, there are model-based annual estimates of bycatch available for both gillnet and bottom trawl fisheries in the Mid-Atlantic (Murray 2009a; Warden 2011a), which provide an estimate of the total number of encounters based on an extrapolation of observed interactions. These analyses only encompass the Mid-Atlantic because there were no interactions with gillnet gear and only one interaction with bottom trawl gear in the Gulf of Maine/Georges Bank area during the time periods used in both analyses. With so few records outside the Mid-Atlantic, too little information was available to support robust model-based analyses for loggerheads throughout the entire action area. Similarly, too few interactions were observed with non-loggerhead sea turtle species throughout the Gulf of Maine, Georges Bank, and Mid-Atlantic to support model-based bycatch estimates for those species in gillnet and bottom trawl gear in any part of the action area (Murray 2009b; Warden 2011b).

The majority of interactions between sea turtles and fisheries considered in this Opinion have occurred south of the Gulf of Maine/Georges Bank; this is likely because 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 off southern New England and in the Mid-Atlantic is coincident with several fisheries which may either target or incidentally land fish species managed under the seven FMPs discussed in this Opinion. As indicated above, the vast majority of sea turtle interactions with the gillnet and trawl components of these fisheries involve loggerheads (Murray 2009a; Warden 2011a).

From 1995-2006, NEFOP observers reported a total of 41 loggerhead, five leatherback, eight Kemp's ridley, five green, and 13 unidentified sea turtles incidentally caught in Mid-Atlantic sink gillnet gear (Murray 2009b). No sea turtle captures in gillnet gear were documented in the Northeast (east of Cape Cod and in the Gulf of Maine) during this time period. The highest estimated bycatch rates of loggerheads in gillnet gear occurred in warm (>15°C), southern Mid-Atlantic waters (south of 36° N) and in large mesh (>17.8 centimeters) gear. Loggerhead bycatch in gillnets occurred in all months except January. Observers reported loggerheads from depths ranging from 1.8-76.8 meters (mean = 28.0 meters) and in waters with SSTs ranging from 8.6°-27.8°C (mean = 17.7°C).

**Table 27** Annual average estimates of and 95% confidence intervals for observed loggerhead sea turtle bycatch in Mid-Atlantic bottom otter trawl gear from 2005-2008 and Mid-Atlantic gillnet gear from 2002-2006, as presented in Warden (2011a) and Murray (2009a), respectively.

| | Bottom Otter Trawl (2005-2008) | | Gillnet (2002-2006) | |
|---|---|---|---|---|
| FMP Group | Mean | 95% CI | Mean | 95% CI |
| Northeast Multispecies | 5 | 3-9 | * | - |
| - Large mesh | 3 | 1-5 | - | - |
| - Small mesh | 3 | 1-4 | - | - |
| Monkfish | 2 | 1-3 | 118 | 68-171 |
| Spiny Dogfish | 0 | 0 | 1 | 0-1 |
| Bluefish | 4 | 3-5 | 48 | 23-79 |
| Skates | 7 | 4-11 | 9 | 5-15 |
| Mackerel/Squid/Butterfish | 25 | 13-37 | * | - |
| Summer Flounder/Scup/ Black Sea Bass | 110** (60 observable; 50 unobservable, quantifiable) | 83-139 (44-77 for observable) | * | - |
| Combined total for all seven FMPs using upper CIs* | | 213 | | 269* |

* Multiple groundfish species in the multispecies complex, as well as Atlantic mackerel, butterfish, scup, and black sea bass are grouped into an "other species" category in Murray (2009a), for which the annual average estimated bycatch of loggerheads attributable to all those species combined is 3 turtles. There is no 95% CI for this estimate threrefore, we added the three turtles to the combined total for all seven FMPs.
** Murray 2011a reports the estimated total Summer Flounder/Scup/Black Sea Bass FMP group interactions to include both observable interactions (i.e. turtles captured in the gear or observed at the surface) and unobservable, quantifiable interactions (i.e. turtles that may have passed through the TED but could not be seen by the observer).

Observers reported 112 loggerhead sea turtle interactions with non-TED bottom otter trawl gear fished in the Mid-Atlantic from 1994-2008 (Warden 2011b). Bottom trawls for fish were involved in 99 of the interactions, while bottom trawls for scallops were involved in the other 13. Observed sea turtle interactions not included in the Warden (2011b) analysis included one loggerhead outside of the Mid-Atlantic. In addition, three Kemp's ridleys, two leatherbacks, and six unidentified sea turtles were taken during this period. Thirteen moderately or severely decomposed carcasses (four loggerheads and nine unidentified) were also excluded as those mortalities were not likely due to the gear interaction. Warden (2011b) found that latitude, depth, and SST were the variables best correlated with the loggerhead interaction rate with bottom trawl gear, with interaction rates being highest south of 37° N latitude in waters <50 meters deep and with SSTs >15°C.

Documented loggerhead interactions with gillnet and bottom trawl gear after the time periods analyzed in Murray (2009b) and Warden (2011b) through 2011 are presented in the table below for additional reference, even though they are not yet included in any model-based estimates of loggerhead bycatch in the Mid-Atlantic. For loggerhead sea turtles, the model-based estimates of annual bycatch in gillnet and bottom trawl gear published in Murray (2009a, 2009b) and Warden (2011a, 2011b) represent the best available information for and analysis of bycatch in the seven fisheries assessed in this Opinion. These estimates are described further in section 7.2.3. Such estimates for gillnet and trawl gear are not available for leatherback, Kemp's ridley, and green sea turtles. Therefore, raw fisheries observer data for these species represent the best available information.

**Table 28** Documented bycatch of loggerhead sea turtles (excluding moderately and severely decomposed sea turtles) in bottom otter trawl gear (fish) from 2009-2011 and gillnet gear from 2007-2011. Gillnet gear includes fixed or anchored sink, drift sink, and drift floating gillnets. Source: NEFOP database.

| | Bottom Otter Trawl (2009-2011) | Gillnet (2007-2011) |
|---|---|---|
| Loggerhead captures | 56 | 7 |

Observer reports from the ASM program, which started in May 2010 and covers the multispecies fishery, have documented an additional seven loggerhead, one leatherback, and one unidentified hard-shell sea turtle interactions with gillnet gear, as well as two leatherback and one unidentified sea turtle interactions with bottom trawl gear through 2011.

**Table 29** Documented bycatch of sea turtles (excluding moderately and severely decomposed sea turtles) in bottom otter trawl (fish and scallops) and gillnet gear recorded during the ASM program in 2010 and 2011. Gillnet gear includes fixed or anchored sink, drift sink, and drift floating gillnets. Source: NEFSC FSB database.

| | Documented # of bycatch in bottom otter trawl gear | Documented # of bycatch in gillnet gear |
|---|---|---|
| Loggerhead sea turtle | 0 | 7 |
| Leatherback sea turtle | 2 | 1 |
| Kemp's ridley sea turtle | 0 | 0 |
| Green sea turtle | 0 | 0 |
| Unidentified sea turtle | 1 | 1 |

While it may be informative to look at the number of leatherback, Kemp's ridley, and green sea turtles observed to have been captured on gillnet or bottom trawl trips when the majority of the landings were any of the species covered under the seven FMPs, using this number as the estimated number of interactions would be an underestimate in two ways. First, sea turtles could have been captured on trips where these species were part of the catch, but constituted less than the majority of the catch. Second, these captures are only observed captures and we are not currently able to extrapolate this number to generate an estimate of total bycatch. In order to partially compensate for this underestimate, for the purposes of estimating interactions of leatherback, Kemp's ridley, and green sea turtles with fishing gear authorized under the seven FMPs assessed in this Opinion, we look at overall interactions by gear type recorded by the NEFOP, regardless of the most landed commercial species. We can then add in any additional sea turtle interactions documented annually through the ASM program to be comprehensive in the inclusion of documented interactions in our estimate.

**Table 30** Documented bycatch of leatherback, Kemp's ridley, green, and unidentified sea turtles (excluding moderately and severely decomposed sea turtles) in bottom otter trawl (BOT: fish) and gillnet gear recorded by the NEFOP from 2002-2011. Source: NEFSC FSB database.

| | **Documented # of bycatch in BOT gear** | **Documented # of bycatch/year in BOT gear** | **Documented # of bycatch in gillnet gear** | **Documented # of bycatch/year in gillnet gear** |
|---|---|---|---|---|
| Leatherback sea turtle | 2 | 0.2 | 4 | 0.4 |
| Kemp's ridley sea turtle | 2 | 0.2 | 6 | 0.6 |
| Green sea turtle | 1 | 0.1 | 14 | 1.4 |
| Unidentified sea turtle | 5 | 0.5 | 7 | 0.7 |

Observations of sea turtle interactions in gillnet and bottom trawl gear indicate that fisheries using these gear types are capable of incidentally capturing sea turtles and that some of these interactions are lethal. 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 free of entangling fishing gear and are thus vulnerable to drowning from forced submergence, although some have been recovered alive in sink gillnets.

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

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

Of the seven fisheries assessed in this Opinion, trap/pot effort occurs mainly in the black sea bass fishery (where it accounts for nearly half the annual landings) and is a minor component of the scup fishery. Trap/pot gear has been associated with sea turtle entanglements in this as well as the comparable trap/pot fisheries for lobster, whelk, crabs, and conch (STDN database). Most of these fisheries use similar gear configurations and fishing methods, including the use of vertical lines and buoy systems which can entangle leatherback and loggerhead sea. Black sea bass and scup trap/pot effort occurs mainly in Southern New England and Mid-Atlantic inshore and offshore waters, where concentrations of leatherbacks, greens, and loggerheads might be expected at certain times of the year. Determining the actual level of sea turtle interactions with black sea bass and scup trap/pot gear set in the action area waters is not possible given the lack of data on the relationship between the concentration of trap/pot gear and the level of entanglement risk for leatherback and loggerhead sea turtles.

Black sea bass and scup trap construction requirements are very similar in the state and federal fisheries, and effort (mostly state) occurs throughout the year but mainly during the spring and late fall. The vast majority of both state and Federal trap/pot fishing effort for black sea bass and scup occurs in the depth range (0-120 feet) where sea turtles are known to occur most frequently; thus, neither fishery is likely to have a disproportionate rate of sea turtle entanglements based on the distributions of sea turtles and black sea bass/scup trap/pot fishing effort. Since the gear, timing, and distribution of effort with respect to sea turtle abundance are essentially the same in both state and federal waters, we believe the number of sea turtle entanglements reported in the state and federal fisheries is the best estimate of sea turtle entanglements.

No method has yet been identified for predicting the number of sea turtle entanglements in the black sea bass and scup trap/pot fisheries. As recorded in the STDN database, leatherback, green, and loggerhead sea turtles have become entangled in trap/pot gear fished in the action area. The black sea bass trap/pot fishery has been confirmed in ten leatherback entanglements from 2002-2011 (STDN database). All interactions with the black sea bass trap/pot fishery have occurred in Massachusetts state waters during the month of August in the following four years: 2003 (1), 2004 (2), 2007 (4), and 2008 (3). The formation of the STDN in 2002 has increased the detail and accuracy of sea turtle entanglement data. Entanglement data may be skewed to show more entanglements in state waters, as these areas are more highly used by boaters who tend to report the majority of entanglements.

In terms of commercial hook and line gear, only the spiny dogfish and multispecies fisheries allocate significant portions of their quotas to these gear types (namely bottom longlines and handlines). Sea turtle bycatch has often been observed in hook and line fisheries, notably the pelagic longline fisheries. Loggerheads and Kemp's ridleys are known to investigate and bite baited hooks according to reports from commercial fishermen fishing for reef fish and sharks with both single rigs and bottom longlines (TEWG 2000; SEFSC 2001). However, no documented interactions of ESA-listed sea turtles have been recorded in the commercial Northeast bottom longline or handline fisheries from 2002-2011 (NEFSC FSB database). Due to the lack of observed interactions in both the spiny dogfish and multispecies hook and line fisheries and because hook and line gear accounts for a small portion of the effort and landings for each fishery (less than 15%), interactions with sea turtles are likely to be extremely rare and unlikely.

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. 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, possibly with embedded/ingested hooks and/or trailing line. Others may be cut free by fishermen and intentionally released. These sea turtles may also have 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 recreational components of the seven fisheries will be an underestimate since analyzable data are only currently available for the commercial component of the fishery.

### 7.2.5 Factors Affecting Atlantic Sturgeon Interactions by Gear Type

Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007). Because of the benthic nature of their invertebrate prey, it is likely that feeding Atlantic sturgeon could swim into or become entangled in sink gillnet gear or be captured by bottom otter trawl gear operating in the action area. Gillnet gear is used by five of the seven fisheries, and bottom trawl gear is used by six of the seven fisheries. It is also possible that bottom longline gear, which is used in three fisheries, could hook Atlantic sturgeon while foraging, but there have been no reported interactions.

While migrating, Atlantic sturgeon may be present throughout the water column and could interact with trawl gear while it is moving through the water column. Atlantic sturgeon interactions with gillnet and bottom trawl gear are likely at times when and in areas where their distribution overlaps with the operation of the fisheries. Atlantic sturgeon also may encounter hooks from both hook-and-line gear and longline gear while traveling through the water column.

A review of all available information resulted in one reported capture of an Atlantic sturgeon in a crab pot in Chesapeake Bay as part of a Reward Program for Maryland. No incidents of trap/pot capture have been reported in any of the fisheries under consultation.

The factors currently thought to be affecting Atlantic sturgeon interactions with fishing gear and mortality due to fishing gear are:

- type of gear
- depth of gear
- location of gear
- mesh size
- soak/tow times
- tie-downs on gillnets
- water temperature
- geographic formations that influence placement of fishing gear and travel routes of sturgeon

The highest incidence of sturgeon bycatch in sink gillnets is associated with depths of <40 meters, larger mesh sizes, and the months April-May (ASMFC TC 2007). Sturgeon bycatch in ocean fisheries is actually documented in all four seasons with higher numbers of interactions in November and December in addition to April and May (NEFSC 2011). Mortality is also correlated to higher water temperatures, the use of tie-downs, and increased soak times (>24 hours) (ASMFC TC 2007). Most observed sturgeon deaths occur in sink gillnet fisheries. For otter trawl fisheries, Atlantic sturgeon bycatch incidence is highest in depths <30 meters and in the month of June.

### 7.2.6 Description of Existing Information on Interactions with Atlantic Sturgeon

Subadult and adult Atlantic sturgeon may be present in the action area year-round. In the marine environment, Atlantic sturgeon are most often captured in depths less than 40 meters. For sink gillnets, higher levels of sturgeon bycatch were associated with depths of less than 40 meters, mesh sizes of greater than 10 inches, and the months of April and May (ASMFC TC 2007). For otter trawl fisheries, the highest incidence of sturgeon bycatch was associated with depths less than 30 meters (ASMFC TC 2007). Atlantic sturgeon captures are reported by observers and are included in the NEFOP database.

We have reviewed available bycatch information and have found that Atlantic sturgeon are frequently reported to interact with both gillnet and trawl gear throughout the action area (Stein *et al.* 2004a; ASMFC TC 2007; NEFSC 2011a). Given the known capture of Atlantic sturgeon in gillnet and trawl fisheries operating in the action area, it is reasonable to anticipate bycatch likely occurs in both the gillnet and trawl components of several, if not all, of the seven fisheries assessed in this Opinion. Discussion of the three studies examining Atlantic sturgeon bycatch and mortality in commercial fishing gear along the East Coast follows.

Stein *et al.* (2004a) investigated fishing records collected by onboard observers for 1989-2000 to calculate Atlantic sturgeon bycatch and mortality for different gear types. The records showed that the highest levels of bycatch occurred in fisheries using sink gillnets (targeting spiny dogfish, monkfish, and Atlantic cod) and that bycatch was higher in the southern parts of the fisheries. The mortality rate for Atlantic sturgeon captured in sink gillnets was 22%, and the peak occurred in winter and spring. Inshore drift gillnets also showed high capture rates for Atlantic sturgeon, peaking in April, and mortality was calculated to be 10%. Otter trawls also accounted for high levels of bycatch, with bycatch peaking in winter and late spring, but there were no observed mortalities. However, the effect of fishing gear may last beyond contact and release (Stein *et al.* 2004a, citing Boreman 1997; Kynard 1997; Caswell *et al.* 1999; Clark and Hare 1998).

Stein *et al.* (2004a) suggested that the following factors may affect bycatch rates:

(1) Differences in regional temperatures that affect movements and migration patterns, thus affecting the amount of time sturgeon spend in the marine environment where fishing is occurring, particularly for the subadult and non-spawning adults.
(2) Geographic formations, such as the narrow continental shelf at the Mid-Atlantic Bight, that affect foraging sturgeon and fishing gear use, bringing them into closer contact.

Stein *et al.* (2004a) also noted that 85% of all recorded sturgeon bycatch involved the following targeted species: monkfish, spiny dogfish, Atlantic cod, summer flounder, American shad and scup. Bycatch was at its lowest in the summer months, when warm waters may force Atlantic sturgeon to seek thermal refuges in estuaries and river systems.

The ASMFC's Technical Committee issued a 2007 report on the estimated bycatch of Atlantic sturgeon in coastal Atlantic commercial fisheries of New England and the Mid-Atlantic using different methodology and a different time frame (2001-2006) than Stein *et al.* used. While not directly comparable, both studies found that deaths were infrequent in the otter trawl observer dataset. The ASMFC report found substantially lower bycatch in both gillnet and otter trawl datasets, and substantially lower mortality in sink gillnets (13.8% as compared to 22%).

It is important to note that observer coverage, on which this data is based, varies across fisheries. However, some patterns did emerge among the factors associated with mortality in sink gillnets: tie-downs, mesh sizes, water temperature, and soak times.

- Larger mesh sizes, particularly the 12-inch mesh, showed high mortality rates
- Longer soak times increased bycatch and mortality
- Warmer water temperatures resulted in higher mortalities
    - In warmer waters, soak times of >24 hours resulted in 40% mortality and soak times of <24 hours resulted in 14% mortality
- Significant positive associations with higher mortalities and warmer water combined with tie-downs, as well as longer soak times combined with tie-downs.

The third study, the NEFSC report (2011b), examined data from the NEFOP and ASM programs collected from 2006 to 2010 in otter trawl and sink gillnet fisheries and expanded the frequency of encounters by using total landings recorded in vessel trip reports.

The NEFSC report also charactarized observed and estimated sturgeon takes by division and quarter, as well as provided annual and total predicted takes and relative influence of FMP species groups to annual take estimates. The fisheries with the highest predicted takes rate using sink gillnet gear were monkfish, skate and flounder/scup/black sea bass. The fisheries with the highest predicted takes rate using otter trawls were flounder/scup/black sea bass, skate, and squid/mackerel/butterfish. The NEFSC study reported a higher rate of Atlantic sturgeon mortality in otter trawls than either of the previous two studies. This study provided interaction rates by mesh sizes in gillnets and trawl gear divided into categories as shown in Table 31.

**Table 31** Atlantic sturgeon interactions with fishing gear recorded in NEFOP and ASM databases for years 2006-2010.

| | % interactions |
|---|---|
| Small mesh trawls ( <5.5 inches) | 26.71 |
| Large mesh trawls ( ≥5.5 inches) | 15.74 |
| Small mesh gillnets ( <5.5 inches) | 7.63 |
| Large mesh gillnets ( 5.5 ≤8.0 inches) | 16.53 |
| Extra large mesh gillnets ( >8.0 inches) | 33.39 |

#### 7.2.7 Factors Affecting GOM DPS Atlantic Salmon Interactions by Gear Type

Atlantic salmon in the ocean are pelagic and highly surface oriented (Kocik and Sheehan 2006; Renkawitz *et al.* 2012). The preferred habitat of post-smolt salmon in the open ocean is principally the upper 10 meters of the water column (Baum 1997, ICES SGBYSAL 2005), although there is evidence of forays into deeper water for shorter periods. Adult Atlantic salmon demonstrate a wider depth profile (ICES SGBYSAL 2005), but overall salmon tend to be distributed in the surface layer, and all fisheries covering this part of the water column are considered to have a potential to intercept salmon.

Due to these factors and the limited abundance of Atlantic salmon, they are not typically caught in the seven fisheries under discussion. Beland (1984) reported that fewer than 100 salmon per year were incidentally caught in commercial fisheries in the coastal waters of Maine.

While migrating, Atlantic salmon may be present throughout the water column and could interact with trawl gear. Atlantic salmon interactions with gillnet and bottom trawl gear are likely at times when and in areas where their distribution overlaps with the operation of the fisheries. Atlantic salmon also may encounter hooks from both hook-and-line gear and longline gear while traveling through the water column.

Gillnet gear is used by five of the seven fisheries, and bottom trawl gear is used by six of the seven fisheries. All observed takes of Atlantic salmon occurred in gillnets or bottom trawls. It is also possible that bottom longline gear, which is used in three fisheries, could hook Atlantic salmon while foraging, but there have been no reported interactions.

#### 7.2.8 Description of Existing Information on Interactions with Atlantic Salmon

Adult Atlantic salmon may be present in the action area year-round, however they are rarely captured in the marine environment. NEFOP data from 1989 through 2011 show records of incidental Atlantic salmon bycatch in 6 of 22 years, with a total of 13 individuals caught. There is no information available on the genetics of these caught Atlantic salmon, so we do not know how many of these salmon are part of the GOM DPS. It is likely that at least some of these salmon, particularly those caught south of Cape Cod, originated from the stocking program in the Connecticut River. The Atlantic salmon caught off the coast of Maine are more likely to be of the GOM DPS. However, as their genetic status is unknown, we will assume for the purposes of this analysis that all 13 are GOM DPS salmon. Reports received through September of 2012 showed no additional incidental catch of Atlantic salmon.

Of the observed incidentally caught Atlantic salmon, eight were listed as "discarded," which is assumed to be a live discard (Kocik, pers comm, Feb 11, 2013). Five of the 13 were listed as mortalities. The incidental takes of Atlantic salmon occurred using sink gillnets (9) and bottom otter trawls (4). Observed captures occurred in November (6), June (3), March (2), April (1) and May (1). The most recent data, from 2004 to 2011, show incidental captures in the multispecies and monkfish fisheries during the spring months in areas offshore (statistical areas 522 and 525) and in the Gulf of Maine (statistical areas 513 and 514).

## 7.3 Anticipated Effects of the Proposed Action

### 7.3.1 Effects to Prey

*ESA-Listed Cetaceans*
We have determined that the continued operation of the seven fisheries will not have any adverse effects on the availability of prey for right, humpback, fin, and sei whales. Right whales and sei whales feed on copepods (Perry *et al.* 1999). The fisheries will not affect the availability of copepods for foraging right and sei whales because copepods are too small to be captured in the fisheries fishing gear. The fisheries will also have no effect on the oceanographic conditions and structures of the Gulf of Maine, Georges Bank, Jordan Basin, Wilkinson Basin, and Georges Basin that contribute to the dense aggregations of late stage and diapausing *Calanus finmarchicus* that attract right and sei whales to this region.

Humpback and fin whales feed on krill as well as small schooling fish (*e.g.,* sand lance, herring, mackerel) (Aguilar 2002; Clapham 2002). The fisheries' fishing gear operates on or very near the bottom. Fish species caught in the fisheries' gear are species that live in benthic habitat (on or very near the bottom) such as flounders. Schooling fish, such as herring and mackerel, occur within the water column, and therefore, with the exception of the mackerel/squid/butterfish fishery, the continued operation of the fisheries will not affect the availability of prey for foraging humpback or fin whales. Although small schooling fish species (including mackerel) may be caught in net gear targeting mackerel/squid/ butterfish, we have found no information that indicates this results in significant impacts to ESA-listed cetaceans.

*ESA-Listed Sea Turtles*
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 seven fisheries under consultation. 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; NMFS and USFWS 1992b; Bjorndal 1997). Therefore, the seven fisheries will not affect the availability of prey for leatherback and green sea turtles in the action area.

Neritic juveniles and adults of both loggerhead and Kemp's ridley sea turtles are known to feed on species that are caught as bycatch in numerous fisheries (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993; Burke *et al.* 1994; Morreale and Standora 2005; Seney and Musick 2005). 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). While the fisheries that target crab species may be impacting loggerheads Kemp's ridleys by reducing available prey the crabs caught as bycatch are expected to be returned to the water alive, dead or injured to the extent that the organisms will shortly die. Injured or deceased bycatch would still be available as prey for sea turtles, particularly loggerheads, which are known to eat a variety of live prey as well as scavenge dead organisms (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993; Morreale and Standora 2005). Kemp's ridley sea turtles have shown increased nesting for the last several years, which strongly suggests that the species is not food limited. The facts that nesting is increasing now and that fishing effort was likely greater during the time that current nesters were maturing suggest that the fisheries are not having any negative effect on Kemp's ridley prey availability.

*Atlantic Sturgeon*
Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007).

Sink gillnets are anchored to the bottom and fish in the lower one-third of the water column. Although sink gillnets are anchored to the seafloor, several studies have found that gillnet gear has little or low impact on bottom habitat (NEFSC 2002; Morgan and Chuenpagdee 2003; GBCHS 2010). Any negative effect from gillnets would vary between fishing habitats, with very low levels of damage on sand, some damage lasting a few days on mud, and more lasting damage on hard bottom clay habitats (NEFSC 2002). Sink gillnets are therefore expected to have discountable effects on Atlantic sturgeon prey.

The effects of bottom trawls on benthic community structure have been the subject of a number of studies. In general, the severity of the impacts to bottom communities is a function of three variables: (1) energy of the environment, (2) type of gear used, and (3) intensity of trawling. High-energy and frequently disturbed environments are inhabited by organisms that are adapted to this stress and/or are short-lived and are unlikely to be severely affected, while stable environments with long-lived species are more likely to experience long-term and significant changes to the benthic community (Stevenson 2004, Mirarchi and CR Environmental 2005, Johnson 2002). Modern otter trawls are lighter than older trawls and scallop dredges, and cause less disturbance to benthic communities, but many older-style beam trawls are still in use (Mirarchi and CR Environmental 2005).The intensity of trawling also affects benthic communities, and significant loss of large sessile epifauna from hard substrates has been demonstrated (Stevenson 2004, Mirarchi and CR Environmental 2005). A majority of studies has found that trawling on mud bottoms decreases the species richness, diversity, abundance, and biomass (Johnson 2002, Stevenson 2004). However, a recent Massachusetts Bay trawling study found no difference between the species composition in trawled and control lanes, but found that faunal density was slightly higher in the trawled lanes (Mirarchi and CR Environmental 2005). While there may be some changes to the benthic communities on which Atlantic sturgeon feed as a result of bottom trawling, there is no evidence the bottom trawl activities of the seven fisheries have a negative impact on availability of Atlantic sturgeon prey.

The trap/pot gear used in the black sea bass and scup fisheries is considered to have low impact to bottom habitat, and is unlikely to incidentally capture Atlantic sturgeon prey. Hook-and-line gear is also unlikely to affect prey, as it has little effect on bottom habitat and is unlikely to incidentally capture Atlantic sturgeon prey. Currently, there is no indication that Atlantic sturgeon are food-limited or that commercial fisheries might negatively impact their food availability, given the diversity of their diets.

*Atlantic Salmon*

Upon completion of the physiological transition to salt water, the post-smolt Atlantic salmon grows rapidly and has been documented to move in small schools loosely aggregated close to the surface (Dutil and Coutu 1988). After entering into the nearshore waters of Canada, the U.S. post-smolts become part of a mixture of stocks of Atlantic salmon from various North American streams. Their diet includes invertebrates, amphipods, euphausiids, and fish (Hislop and Youngson 1984; Jutila and Toivonen 1985; Fraser 1987; Hislop and Shelton 1993). Results from a 2001-2005 post-smolt trawl survey in Penobscot Bay and the nearshore waters of the Gulf of Maine indicate that Atlantic salmon post-smolts are prevalent in the upper water column (Sheehan *et al.* 2005).

Most of the GOM DPS-origin salmon spend two winters in the ocean before returning to streams for spawning. Aggregations of Atlantic salmon may still occur after the first winter at sea, but most evidence indicates that they travel individually (Reddin 1985). At this stage, Atlantic salmon primarily eat fish, feeding upon capelin, herring, and sand lance (Hansen and Pethon 1985; Reddin 1985; Hislop and Shelton 1993).

The fisheries' fishing gear operates on or very near the bottom. Fish species caught in the fisheries' gear are species that live in benthic habitat (on or very near the bottom) such as flounders. Schooling fish, such as herring, capelin and sand lance, occur within the water column, and therefore, with the exception of the mackerel/squid/butterfish fishery, the continued operation of the fisheries will not affect the availability of prey for foraging post-smolt and adult Atlantic salmon. Although small schooling fish species (including mackerel) may be caught in net gear targeting mackerel/squid/ butterfish, we have found no information that indicates this results in significant impacts to the GOM DPS of Atlantic salmon.

#### 7.3.2 Effects to Habitat

Of all the gears used in the seven fisheries, bottom trawl is the only gear type that has the potential to adversely affect bottom habitat in the action area (NMFS 2003a). A panel of experts has previously concluded that the effects of even light weight otter trawl gear would include: (1) scraping or plowing of the doors on the bottom, sometimes creating furrows along their path; (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom; (3) removal or damage to benthic or demersal species; and (4) removal or damage to structure forming biota. The panel also concluded that the greatest impacts from otter trawls occur in high and low energy gravel habitats and in hard clay outcroppings, and that sand habitats were the least likely to be impacted (NREFHSC 2002).

Alterations of bottom habitat should not affect foraging right, humpback, fin, and sei whales (Baumgartner *et al.* 2003; IWC 1992; Pace and Merrick 2008; Perry *et al.* 1999), but they may be temporarily disturbed by the use of bottom fishing gear.

The foraging distribution of Kemp's ridley, loggerhead, and green sea turtles in Mid-Atlantic and New England waters as far north as approximately Cape Cod, do not typically occur in gravel habitats. Leatherback sea turtles have a broader distribution in New England waters, which may include clay outcroppings, but are pelagic feeders, which should be less impacted by alterations to benthic habitat. For these reasons, and the lack of any evidence that fishing practices affect habitats in degrees that harm or harass ESA-listed species, NMFS finds that while continued fishing efforts by the fisheries may potentially alter benthic habitats, these alterations will be insignificant to ESA-listed sea turtles.

Atlantic sturgeon use the action area as a migratory route and for overwintering and foraging. Any effects on habitat due to bottom trawl gear are most likely to be on Atlantic sturgeon prey items, as discussed above. Atlantic sturgeon are known to aggregate in certain areas and at certain times of the year, and some of these areas experience high fishing effort. While the reason for the aggregations is currently unknown, it is suspected that they aggregate at the mouths of large rivers for foraging in the summer and in areas off the New York Bight and off North Carolina in the winter. Despite the overlap in aggregations with some areas of high fishing effort, we have no information that indicates negative effects on Atlantic sturgeon prey items, although foraging, overwintering, and migrations may be temporarily disturbed by the use of bottom fishing gear. Gillnet gear may also impede Atlantic sturgeon migrations, but the effects are also expected to be insignificant, unless entanglement results, as discussed below (see section 7.2.5).

Atlantic salmon also use the action area as a migratory route and for foraging. The effects on habitat due to bottom trawl gear are most likely to affect some Atlantic salmon prey items, as discussed above. Aggregations of Atlantic salmon may occur both at the post-smolt stage and after their first winter at sea, but most evidence indicates that they travel individually as adults (Reddin 1985). Foraging and travel activity may be temporarily disturbed by the use of bottom fishing gear, but the effects are expected to be insignificant. Gillnet gear may also impede Atlantic salmon travel, but the effects are also expected to be insignificant, unless entanglement results, as discussed below.

### 7.3.3 Vessel Strikes

*ESA-Listed Cetaceans*

Vessel strikes are a threat to a number of marine species worldwide including ESA-listed large whales. Vessel collisions with marine mammals can result in death by massive trauma, hemorrhaging, broken bones, and propeller wounds (Knowlton and Kraus, 2001; Campbell-Malone, 2007). When large whale species and large vessels are involved, the stricken whale can occasionally be found draped across the ship’s bulbous bow when it arrives in port. Massive propeller wounds can be immediately fatal. However, if relatively superficial, some individuals can recover from seemingly serious collisions, as evidenced by photographic time series of deep lacerations healing on individual animals (Silber *et al.* 2009). Vessel strikes of large whales are a growing problem internationally (Van Waerebeek and Leaper 2008), particulary where endangered or depleted species are involved. A contributing factor is the increase in maritime commerce, which is expected to nearly double over the next 15 years in U.S. ports (U.S. Department of Transportation 2008).

A 2003 report from the NOAA’s Large Whale Ship Strike Database found that only four (3%) of 134 reported incidents (1975-2002) where the type of vessel was known were fishing vessels. Analysis of the ship strike database indicates vessel types faster and/or larger than fishing vessles are more likely to be involved in large whale ship strikes. Injuries and mortalities from vessel strikes are a serious threat to North Atlantic right whales. Based on photographs of catalogued animals from 1935 through 1995, Hamilton *et al.* (1998) estimated that 6.4% of the North

Atlantic right whale population exhibits signs of injury from vessel strikes. Reports received from 2006 to 2010 indicate that right whales experienced five ship strike mortalities and one serious injury (Henry *et al.* 2012). In 2006 alone, four reported mortalities and one serious injury resulted from right whale ship strikes (Henry *et al.* 2012).

Injuries and mortalities from vessel strikes are also a threat to humpback, fin, and sei whales. Vessel strikes accounted for an annual average of 1.4 humpback whale SI/Ms in U.S. waters per year between 2005 and 2009 (Waring *et al.* 2012). The annual average whale vessel strikes in U.S. waters was 1.4 and 0.6 for fin and sei whale respectively.

The effects of vessel strikes on North Atlantic right whales is being addressed by a Ship Strike Reduction Program, but the operational measures are expected to reduce the incidence of ship strike on other large whales to some degree. For more information, see Section 5.4.6.

*ESA-Listed Sea Turtles*
Interactions between vessels and sea turtles occur and can take many forms, from the most severe (death or bisection of an animal or penetration to the viscera), to severed limbs or cracks to the carapace which can also lead to mortality directly or indirectly. Sea turtle stranding data for the U.S. Gulf of Mexico and Atlantic coasts, Puerto Rico, and the U.S. Virgin Islands show that between 1986 and 1993, about 9% of living and dead stranded sea turtles had propeller or other boat strike injuries (Lutcavage *et al.* 1997). According to 2001 STSSN stranding data, at least 33 sea turtles (loggerhead, green, Kemp's ridley and leatherbacks) that stranded on beaches within the northeast (Maine through North Carolina) were struck by a boat. This number underestimates the actual number of boat strikes that occur since not every boat struck turtle will strand, every stranded turtle will not be found, and many stranded turtles are too decomposed to determine whether the turtle was struck by a boat. It should be noted, however, that it is not known whether all boat strikes were the cause of death or whether they occurred post-mortem (NMFS SEFSC 2001).

Information is lacking on the type or speed of vessels involved in turtle vessel strikes. However, there does appear to be a correlation between the number of vessel struck turtles and the level of recreational boat traffic (NRC 1990). Although little is known about a sea turtle's reaction to vessel traffic, it is generally assumed that turtles are more likely to avoid injury from slower-moving vessels since the turtle has more time to maneuver and avoid the vessel. In addition, the risk of ship strike will be influenced by the amount of time the animal remains near the surface of the water.

*Atlantic Sturgeon*
Based on the best available information, we have concluded that vessel strikes are a significant threat to Atlantic sturgeon (77 FR 5880 and 77 FR 5914; February 6, 2012). Given that Atlantic sturgeon subadults and adults from all DPSs use ocean waters from Labrador, Canada to Cape Canaveral, FL, as well as estuaries of large rivers along the U.S. East Coast, activities affecting these water bodies are likely to impact more than one Atlantic sturgeon DPS.

The exact number of Atlantic sturgeon killed as a result of being struck by boat hulls or propellers is unknown. The factors relevant to determining the risk to Atlantic sturgeon from vessel strikes are currently unknown, but may be related to size and speed of the vessels, navigational clearance (i.e., depth of water and draft of the vessel) in the area where the vessel is operating, and the behavior of Atlantic sturgeon in the area (e.g., foraging, migrating, etc.). While we have some information on the number of mortalities in the Delaware and James rivers that are thought to be due to vessel strikes (see Status of the Species, Section 4.4), we are not able to use those numbers to extrapolate effects throughout one or more DPS. This is because of (1) the small number of data points and, (2) lack of information on the percent of incidences that the observed mortalities represent. While vessel strikes are believed to be a threat in several rivers as noted in the Status of the Species section above, we do not have information that suggests that Atlantic sturgeon are struck by vessels in the open marine environment of the action area. Given the depths in which most of these fisheries are prosecuted and the depths at which Atlantic sturgeon are most likely migrating in the ocean, vessel strikes in the action area are most likely very uncommon.

*Atlantic Salmon*
The threats assessment done for Atlantic salmon as part of the 2009 endangered listing of the expanded GOM DPS did not list vessel strikes as a high priority threat 74 FR 29344 June 19, 2009). There is no data currently available on vessel strikes and Atlantic salmon.

## 7.4 Anticipated Interactions with Cetaceans

To date, no method has been identified for predicting the level of overall or species-specific cetacean bycatch in the seven fisheries. Some whale mortalities may never be observed or reported, thus the actual annual number of documented mortalities are likely a subset of the actual number of entanglement related mortalities that occur. Additionally, assignment of a specific fishery to a reported entanglement is rarely possible because even in those rare cases where gear is retrieved, identification remains problematic because the same gear (*e.g.,* lines and webbing) is used in multiple fisheries.

The analysis of entanglement events used in this Opinion differs in an important way from the reporting in the NOAA Stock Assessment Reports for Marine Mammals. Specifically, gear analyses were the criteria used to categorize entanglement events to U.S., Canadian, or undefined origin in this Opinion; in contrast, the NOAA Stock Assessment Reports for Marine Mammals initially use the location the animal was first sighted to categorize the events to "U.S. waters" or "Canadian waters," then re-assign any events when/if gear analyses 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 NOAA Stock Assessment Reports for Marine Mammals is to report 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 continuation of the seven fisheries. Thus, for the purposes of this Opinion, NMFS has included entanglement events that have been attributed to gear used in Canadian fisheries as a portion of the the Environmental Baseline and not include them as part of the analysis of impacts of the proposed action because they are not the result of the action under consultation, and in turn, we focus on entanglement events that are of undetermined origin or confirmed U.S. origin since these events are directly attributed to U.S. fisheries or cannot be ruled out as resulting from U.S. fisheries, including those considered in this Opinion. By including gear of "unknown" origin, which may in fact be foreign gear, we are taking a more conservative approach than we would be if we excluded all gear that could not be identified as U.S. origin. 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)].

### 7.4.1 Anticipated Interactions with Cetaceans by Gear Type

#### *7.4.1.1 Otter Trawls*

Right, humpback, fin, and sei whales are not expected to be affected by the use of bottom otter trawl gear given that these large cetaceans have the speed and maneuverability to get out of the way of oncoming mobile gear, including trawl gear. There have been no documented interactions of right, humpback, fin or sei whales with bottom otter trawl gear. Given there are no changes proposed to the fishing practices of the seven fisheries, it is reasonable to anticipate that no interactions of large whales with otter trawl gear will occur in the future.

#### *7.4.1.2 Sink Gillnets and Trap/Pot*

*North Atlantic Right Whales*

From 2006 to 2010, the average annual reported mortality or serious injury to right whales in U.S. waters due to fishery entanglement was 1.6 (Waring *et al.* 2012). Documented entanglements most likely underestimate the extent of the entanglements since not all entanglements are likely to be reported. Consequently, the total level of interaction between fisheries and right whales is unknown. However, studies have estimated that more than 60% of right whales exhibit scars consistent with fishery interactions. Broad based gear modifications developed under the ALWTRP are expected to reduce the number and severity of right whale entanglements.

Between 2006 and 2010, 28 entangled right whales were reported, 27 of which were gillnet, trap/pot, or unknown gear. Of these 27, one case was identified as gillnet gear, three cases identified as trap/pot gear, and 23 cases with unknown gear. Of the entanglements that resulted in serious injury or mortality, four had unknown gear, one had unknown gillnet gear, and two had unknown pot/trap gear. In this time period, approximately 21% of all the reported right whale entanglements resulted in serious injury or mortality (NMFS NERO 2012).

Entanglements of right whales in gillnet and trap/pot gear continue to occur despite the measures implemented by the ALWTRP. The ALWTRP has recently added new measures affecting gillnet and trap/pot gear in the Northeast U.S. While the measures of the ALWTRP are expected to reduce the lethal effect of gillnet and trap/pot gear on right whales, based on the observed range of reported entanglements over the past ten years, the seven fisheries have the potential to seriously injure or kill zero to three right whales per year. The seven fisheries continue to pose a risk of entanglement for North Atlantic right whales.

*Humpback whales*

Between 2006 and 2010, 86 humpback whale entanglements were documented. Six of those entanglements were in gillnet gear, averaging 1.2 per year (NMFS NERO 2012). From 2006 to 2010, there was one documented humpback mortality as a result of entanglement in gillnet gear. Additionally, 19 of the humpback entanglements from 2006-2010 in undocumented gear types resulted in serious injury or mortality. Although there were no documented entanglements of humpback whales in trap/pot gear from any of the seven fisheries in this Opinion, humpback entanglements have been recorded in the American lobster fishery. Since the scup/black sea bass trap/pot fishery uses similar gear that is used in the American lobster fishery, it is possible that humpbacks may become entangled in trap/pot gear set by the seven fisheries. Because serious injuries or mortalities of humpbacks in gillnet and trap/pot gear have occurred in the past, based on the observed range of reported entanglements over the past ten years, we expect that the seven fisheries have the potential to seriously injure or kill zero to eight humpback whales per year.

*Fin whales*

Fin whales are vulnerable to entanglement in gillnet and trap/pot gear while foraging and migrating in areas where gear is present. Entanglements of fin whales have been documented but are considered to occur at a level approaching zero mortality and serious injury rate. From 2006-2010, no fin whales were documented entangled in gillnet or trap/pot gear. However, in that time period there were 12 events where the gear was not identified or recovered and it is possible that some of that gear originated from the seven fisheries (NMFS NERO 2012).

Although some entangled whales may be freed of gear (either by their own actions or with the assistance of the disentanglement network), given the limited survey coverage in the action area, the limited observer coverage in the seven fisheries, that gear is not continuously tended, the logistical difficulties of disentanglement efforts in offshore areas, and the known serious injury or mortality of other whales resulting from gillnet and trap/pot gear, we assume that in the future, based on the observed range of reported entanglements over the past ten years, fin whales may be entangled in gillnet and trap/pot gear and that zero to three entanglements may be detected that result in serious injury or mortality per year.

*Sei whales*
From 2006 to 2010, there were three documented cases of sei whales entangled with unidentified gear; no entanglements have occurred in gear that was identified as gillnet or trap/pot gear. While interactions with sei whales are possible, this species does not frequent inshore waters and therefore is not likely to encounter gillnet or trap/pot gear. Based on documented entanglements, the average annual rate of sei whale entanglements is approximately 0.6. No sei whale mortalities have been reported as a result of entanglement in gillnet or trap/pot gear (NMFS NERO 2012), although it is possible. Based on the observed range of reported entanglements over the past ten years, zero to two serious injury and mortalities due to entanglement of sei whales may be detected per year.

#### *7.4.1.3 Hook Gear*

According to the NMFS analysis of gear interactions with large whales in the Atlantic Ocean, there have been seven humpback interactions with hook and line gear (Table 24). Over the five-year period of data reference (*i.e.*, 2006-2010) there was an annual mean interaction rate of 1.4. The fish targeted in the hook and line interactions have not been determined. Interactions with hook and line gear and right, fin, and sei whales have not been reported. The seven hook and line interactions with humpback whales were observed on live animals and were known to have not been lethal at the time of observation.

The have been no reported large whale interactions with hook and line gear in the action area that has resulted in SI/M. Considering the recent reductions in fishing effort in the multispecies fishery as a result of management efforts (*i.e.*, Amendment 13, Amendment 16) NMFS anticipates an annual rate of future hook and line gear interactions of 1.4 whales detected per year to be a conservatively high estimate; we expect that these whales could be right, humpback, fin or sei whales. None of these interactions are expected to result in SI/M.

## 7.5 Anticipated Interactions with Sea Turtles

As described earlier in this Opinion, the Murray (2009a) and Warden (2011a) reports analyze fishery observer data and VTR data from fishermen in order to estimate the average annual number of sea turtle interactions in gillnet and bottom trawl gear in the Mid-Atlantic that occurred over certain time periods (2002-2006 for gillnets, 2005-2008 for trawls). Unfortunately, these reports are only able to compute bycatch estimates for loggerheads, due to small sample sizes of observer records and a low frequency of encounters for leatherback, Kemp's ridley, and green sea turtles. These reports on Mid-Atlantic interactions represent the most accurate predictor of annual loggerhead sea turtle interactions in the fisheries, as interactions on Georges Bank and in the Gulf of Maine are highly infrequent and have not been able to be assessed statistically. For the other three species of sea turtles, observer reports from the NEFOP and ASM databases represent the best available information on annual bycatch in these fisheries. For trap/pot gear interactions with sea turtles, entanglement data from the STDN represents the best available information on annual bycatch in the black sea bass/scup trap/pot fishery. Interactions with commercial hook and line gear are expected to be rare, and are thus not addressed in this section.

### 7.5.1 Anticipated Interactions of Sea Turtles by Gear Type

The loggerhead sea turtle bycatch estimate methods for gillnets and trawls (Murray 2009a; Warden 2011a) assigned trips and associated bycatch to FMPs or individual species landed based on the distribution of landings for that trip. Trips in a certain time and area using gillnets were estimated to have a certain bycatch rate of loggerhead sea turtles (based on the observed interactions). In the estimate, the gillnet trip and its associated interactions (calculated using the bycatch rate), were assigned to multiple fisheries in a ratio that reflected the catch composition of that trip by weight. This method is meant to reflect that many of the fisheries that operate throughout the Mid-Atlantic region land several species on any given trip.

There are no total bycatch estimates for leatherback, Kemp's ridley, or green sea turtles in gillnet or trawl gear. The very low number of observed non-loggerhead interactions in gillnet and trawl gear suggests that interactions with these species within the action area are even more rare than loggerhead interactions. However, given the fact that observer coverage in these fisheries is less than 100%, it is likely that interactions with non-loggerhead sea turtles have occurred but were not observed or reported. Given effort in the fisheries as a whole, and the seasonal overlap in distribution of these species with operation of gillnet and trawl gear, leatherback, Kemp's ridley, and green sea turtles are likely to interact with both gear types.

***Gillnets***

From 2002 to 2006, the average annual bycatch estimate of loggerheads in Mid-Atlantic sink gillnet gear was 288 turtles (Murray 2009a). For the seven fisheries assessed in this Opinion, the annual average estimates of loggerhead interactions with sink gillnet gear used in those fisheries are presented in Table 28. With the respective 95% CIs, it would be expected that anywhere from the low end to the high end of loggerheads could interact with the gear annually and that would be within the range of estimated interactions based on past records. These estimates of loggerhead sea turtle interactions with Mid-Atlantic sink gillnet gear provide the best available information for determining the anticipated bycatch of loggerhead sea turtles in that gear type in the action area. For this Opinion, we used the upper ends of the 95% CI and therefore estimate an annual average of up to **269** loggerhead sea turtles per year (the upper ends of each 95% CI for monkfish, spiny dogfish, bluefish, and skates plus the 3 loggerheads attributed to the "other species" category added together; 171+1+79+15+3=269) is the best available information on the anticipated number of interactions in the gillnet component of these Northeast fisheries. This represents the total number of interactions expected annually in the gillnet component of these fisheries and not just the number that may be observed. We further believe that any loggerhead interactions in gillnet gear that occur outside of the Mid-Atlantic will be subsumed within this estimate as the upper ends of the 95% CIs (rather than the means) were used.

As summarized in Table 30, the annual average number of leatherback captures in gillnet gear in the action area documented through the NEFOP is 0.4. Since the capture of a partial sea turtle is not possible, we round that number up (as well as all fractions of sea turtles in this Opinion) to one per year. Adding an additional turtle to account for the possibility that the 0.7 unidentified sea turtles (also rounded up to one) captured annually in gillnet gear, as recorded by the NEFOP, could all be leatherbacks gives a total of two captures in gillnet gear annually. Finally, adding in the 0.5 leatherbacks (rounded up to one) and 0.5 unidentified sea turtles (rounded up to one) captured annually, as documented through the ASM program (Table 29), gives us the annual documented capture of **four** leatherback sea turtles in gillnet gear used in the fisheries in this consultation.

The average annual average number of Kemp's ridley captures in gillnet gear in the action area documented through the NEFOP is 0.6. Again, since the capture of a partial sea turtle is not possible, we round that number to one per year. Adding an additional turtle to account for the possibility that the 0.7 unidentified sea turtles (also rounded up to one) captured annually in gillnet gear, as recorded by the NEFOP, could be a Kemp's ridley gives a total of two captures in gillnet gear annually. Finally, adding in the 0.5 unidentified sea turtles (rounded up to one) captured annually, as documented through the ASM program, gives us the annual documented capture of **three** Kemp's ridley sea turtles in gillnet gear used in the fisheries in this consultation.

The average annual number of green sea turtle captures in gillnet gear in the action area documented through the NEFOP is 1.4. Since the capture of a partial sea turtle is not possible, we round that number to two per year. Adding an additional turtle to account for the possibility that the 0.7 unidentified sea turtles (also rounded up to one) captured annually in gillnet gear, as recorded by the NEFOP, could be a green sea turtle gives a total of three captures in gillnet gear annually. Finally, adding in the 0.5 unidentified sea turtles (rounded up to one) captured annually, as documented through the ASM program, gives us the annual documented capture of **four** green sea turtles in gillnet gear used in the fisheries in this consultation.

***Bottom Trawls***

The estimated average annual bycatch of observable loggerhead sea turtles in bottom otter trawl gear during 2005-2008 is 292 turtles with a 95% CI for the annual average of 221-369, with an additional unobservable, but quantifiable 61 turtles excluded by TEDs (95% CI: 41-83) (Warden 2011b). For the seven Northeast fisheries assessed in this Opinion, the annual average estimates of loggerhead interactions with trawl gear used in those fisheries are presented inTable 28. These estimates of loggerhead sea turtle bycatch in bottom otter trawl gear provide the best available information for determining the anticipated number of loggerhead sea turtle interactions per year in the bottom trawl components of these fisheries. For this Opinion, we used the upper end of the 95% CI and therefore estimate that an annual average of up to **213** loggerhead sea turtles (9+5+4+3+0+5+11+37+139=213) is the best available information on the anticipated number of interactions in the bottom trawl component of these fisheries. This represents the total number of interactions we are expecting annually in the bottom trawl component of these fisheries and not just the number observed. We further believe that any interactions in bottom trawl gear that occur outside of the Mid-Atlantic will be subsumed within this estimate (which is the result of the upper ends of the 95% CIs being summed rather than the means).

As summarized in Table 30, the annual average number of leatherback captures in bottom trawl gear in the action area documented through the NEFOP is 0.2. Since the capture of a partial sea turtle is not possible, we round that number up to one per year. Adding an additional turtle to account for the possibility that the 0.5 unidentified sea turtles (also rounded up to one) captured annually in bottom trawl gear, as recorded by the NEFOP, could be a leatherback gives a total of two captures in bottom trawl gear annually. Finally, adding in the 1.0 leatherbacks (two captures over two years) and 0.5 unidentified sea turtles (rounded up to one) captured annually, as documented through the ASM program, gives us the documented annual capture of four leatherback sea turtles in bottom trawl gear used in the seven fisheries in this consultation.

The average annual average number of Kemp's ridley captures in bottom trawl gear in the action area documented through the NEFOP is 0.2. Again, since the capture of a partial sea turtle is not possible, we round that number to one per year. Adding an additional turtle to account for the possibility that the 0.5 unidentified sea turtles (also rounded up to one) captured annually in bottom trawl gear, as recorded by the NEFOP, could be a Kemp's ridley gives a total of two captures in bottom trawl gear annually. Finally, adding in the 0.5 unidentified sea turtles (rounded up to one) captured annually, as documented through the ASM program, gives us the annual documented capture of **three** Kemp's ridley sea turtles in bottom trawl gear used in the seven fisheries in this consultation.

The average annual number of green sea turtle captures in bottom trawl gear in the action area documented through the NEFOP is 0.1. Since the capture of a partial sea turtle is not possible, we round that number to one per year. Adding an additional turtle to account for the possibility that the 0.5 unidentified sea turtles (also rounded up to one) captured annually in bottom trawl gear, as recorded by the NEFOP, could be a green sea turtle gives a total of two captures in bottom trawl gear annually. Finally, adding in the 0.5 unidentified sea turtles (rounded up to one) captured annually, as documented through the ASM program, gives us the annual documented capture of **three** green sea turtles in bottom trawl gear used in the seven fisheries in this consultation.

***Trap/Pot Gear***

The following describes the data used, the processes, and the results of NMFS's analyses for estimating the number of annual sea turtle interactions with the trap/pot component of the black sea bass/scup fishery. When calculating the interaction rates for both leatherback and loggerhead sea turtles, we used STDN vertical line stranding and entanglement records documented during 2002-2011 in state and Federal waters. We believe this approach utilizes the best available information and is the most reasonable as these two species of sea turtles occur throughout the action area, are highly migratory, and can be found in both state and federal waters.

An annual estimate of sea turtle interactions was determined based on the number of confirmed entanglement reports from 2002-2011. As noted above, confirmed leatherback entanglements in black sea bass and scup trap/pot gear have only been reported in state waters. However, the fishery and leatherbacks overlap in both state and Federal waters and we believe that interactions are equally likely in both areas. We, therefore, will take the state waters count of the sea turtle interactions (we only have state entanglements) and apply this to the overall fishery managed under the FMP.

For this Opinion, we will utilize the highest number of annual documented leatherback entanglements per year between 2002-2011 that have been confirmed as attributable to the trap/pot component of the black sea bass and scup fishery as our estimate of annual interactions. The highest number of leatherback sea turtle interactions per year (four) occurred in 2007. Although the actual number of leatherbacks entangled in trap/pot gear per year may be larger, it cannot be extrapolated from the existing STDN data. As a result, we have determined that the maximum number of annual interactions between 2002-2011 represents the best available information on the number of leatherback interactions anticipated in the trap/pot component of the fishery annually. Therefore, we anticipate **four** leatherback interactions annually in trap/pot gear used in the black sea bass/scup fishery.

As previously stated, documentation of loggerhead sea turtle interactions with black sea bass or scup trap/pot gear has not occurred. Using the STDN data, there has only been one documented case of a loggerhead entangled in vertical line gear in the area from Maine to New York from 2002-2011, where the black sea bass and scup trap/pot fisheries are executed. This event was classified as probable and the gear on the animal was not identified. During this time period there were 12 confirmed reports of loggerheads entangled in vertical line gear in other areas, 11 in Virginia and one in New Jersey. Despite the limited reported interactions of loggerheads with trap/pot gear, the possibility exists that interactions will occur. We realize that more loggerheads might be entangled than are actually reported. However, there is not information available to estimate these; therefore, we anticipate **one** loggerhead sea turtle interaction annually in trap/pot gear used in the black sea bass/scup fishery.

#### 7.5.2 Age Classes of Sea Turtles Anticipated to Interact with Each Gear Type

*Loggerhead sea turtles.* The 2008 recovery plan identifies five life stages for loggerhead sea turtles: (1) hatchling: 4 centimeters CCL, 1-5 days; (2) post-hatchling: 4-6 centimeters CCL, <6 months; (3) oceanic juvenile: 8.5-64 centimeters CCL, 7-11.5 years; (4) neritic juvenile: 46-87 centimeters CCL, 13-20 years; and (5) adult male/female: >83 centimeters CCL and >87 centimeters CCL (respectively), >25 years for females (NMFS and USFWS 2008). Both Murray (2009b) and Warden (2011b) presented data on loggerhead sea turtles interacting with gillnet and trawl gear that we can use to determine estimated sizes of future interactions. Sizes of observed loggerheads caught in Mid-Atlantic gillnet gear from 1995-2006, for which measurements could be taken, ranged between 52 and 101 centimeters CCL with a mean of 65.3 centimeters CCL (n=12 turtles) (Murray 2009b). Ten of the 12 (83%) loggerheads measured were under 72 centimeters CCL, a size considered to be within the juvenile life stage (NMFS and USFWS 2008; Murray 2009b). Size classes of loggerheads observed captured in Mid-Atlantic trawl gear between June 1994 and December 2008 spanned both juvenile and adult life stages, although the vast majority (approximately 90%) were juveniles (Warden 2011b). Based on these observer measurements and the known distribution of loggerhead sea turtles captured in other U.S. Atlantic coastal fisheries, we expect that both juvenile and adult loggerheads may be captured in gear used by these seven fisheries because both life stages are present within the action area.

*Leatherback sea turtles*. Sighting and stranding records suggest that both juvenile and adult leatherbacks occur within the action area where the fisheries operate (NMFS and USFWS 1992b; SEFSC 2001). Satellite-tracking of tagged leatherbacks also demonstrates the movement of sexually mature leatherbacks over

U.S. continental shelf waters (James *et al.* 2005a, 2005b). Therefore, both juveniles and adults could interact with these fisheries since both age classes occur in areas where the fisheries operate.

*Kemp's ridley sea turtles*. The post-hatchling stage for Kemp's ridley sea turtles was defined by the TEWG as Kemp's ridleys of 5-20 centimeters standard carapace length (SCL), while turtles 20-60 centimeters SCL were considered to be benthic immature (TEWG 2000). The latter stage is described as sea turtles that have recruited to coastal benthic habitat. Mid-Atlantic and coastal New England waters (as far north as approximately Cape Cod) are known to be developmental foraging habitat for immature Kemp's ridley sea turtles, while adults have been documented from waters and nesting beaches along the South Atlantic coast of the U.S. and in the Gulf of Mexico (Musick and Limpus 1997; TEWG 2000; Morreale and Standora 2005). Given the life history of the species, we expect that only juvenile Kemp's ridley sea turtles are likely to interact with gear used in these fisheries.

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

### 7.5.3 Estimated Mortality of Sea Turtles that Interact with Each Gear Type

Sea turtle interactions with gillnet, bottom trawl, and trap/pot gear likely result in a higher level of sea turtle mortality than is evident based on the number of sea turtles returned to the water alive. Injuries suffered by sea turtles interacting with these gear types 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 to sea turtles affected by the continued operation of these fisheries. It should be noted that the severity of sea turtle submergence injuries as a result of trawl gear interactions will likely be less if the turtle is interacting with a trawl equipped with a TED rather than a trawl without one.

Sea turtles forcibly submerged in any type of restrictive gear eventually suffer fatal consequences from prolonged anoxia and/or seawater infiltration of the lung (Lutcavage *et al.* 1997). A study examining the relationship between tow time and sea turtle mortality in the shrimp trawl fishery showed that mortality was strongly dependent on trawling duration, with the proportion of dead or comatose sea turtles rising from 0% for the first 50 minutes of capture to 70% after 90 minutes of capture (Henwood and Stuntz 1987). However, metabolic changes that can impair a sea turtle's ability to function can occur within minutes of a forced submergence. Most voluntary dives appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status. The story is quite different, however, in forcibly submerged sea turtles, where oxygen stores are rapidly consumed, anaerobic glycolysis is activated, and acid-base balance is disturbed, sometimes to lethal levels (Lutcavage and Lutz 1997). Forced submergence of Kemp's ridley sea turtles in shrimp trawls resulted in an acid-base imbalance after just a few minutes (times that were within the normal dive times for the species) (Stabenau *et al.* 1991). Conversely, recovery times for acid-base levels to return to normal may be prolonged. Henwood and Stuntz (1987) found that it took as long as 20 hours for the acid-base levels of loggerhead sea turtles to return to normal after capture in shrimp trawls for less than 30 minutes. This effect is expected to be worse for sea turtles that are recaptured before metabolic levels have returned to normal.

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

Tows by trawl vessels are usually around one to two hours in duration. However, Murray (2008) found that tow times of bottom otter trawl gear that resulted in sea turtle bycatch ranged from 0.5 to over 5 hours. Shortened tow durations in some fisheries, which have been used to limit large amounts of non-target fish species bycatch, should help to reduce the risk of death from forced submergence for sea turtles caught in trawls, but they do not eliminate the risk. For trawl fisheries, assuming that the mortality rate for sea turtles from forced submergence 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, especially if they are caught at the beginning of long tows.

There are far fewer studies on the effects of forced submergence in gillnets than there are for trawls. However, the risk of a sea turtle drowning as a result of entanglement in gillnet gear is assumed to be greater compared to trawl gear, as gillnets are often left to soak for extended periods of time (*i.e.*, days rather than hours) and are usually anchored to the seafloor. If a sea turtle is caught in a gillnet soon after it is set and is unable to surface for air, the likelihood of mortality is high, as a fisherman may not be back to retrieve it for several days. Soak times for gillnets in which live sea turtles were captured from 1995-2006 ranged between 0.6 and 96 hours (mean = 29.6 hours), and between 22.2 and 216 hours (mean = 80 hours) for gillnets in which fresh dead sea turtles were captured (Murray 2009b).

<u>Serious injury/mortality calculations for gillnet and trawl gear</u>
Until recently, the best available information on loggerhead mortality was the number of dead loggerhead sea turtles documented by the NEFOP and reported in the bycatch estimates (Murray 2008, 2009a). Based on the descriptions provided by fisheries observers, it seems probable that some injured sea turtles observed captured in commercial fishing gear and that were returned to the water alive would have subsequently died as a result of those injuries. We recognized the need to expand guidance originally developed for the scallop dredge fishery to attempt to encompass other Northeast Region gear types (*e.g.*, gillnet, trawl) and a wide range of sea turtle injuries, and to use a consistent approach for assessing post-release survival.

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

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

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

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

The seven fisheries assessed in this Opinion primarily use sink gillnet and bottom otter trawl gear. After the review of observer records from 2006 to 2010, the Northeast sea turtle injury workgroup calculated a resulting mortality rate for gillnet gear of 58% (29 records reviewed). For trawl gear (97 records reviewed), the resulting mortality rate for observable interactions was 47% (Upite *et al.* 2012). Thus, of the 269 loggerhead interactions expected to occur annually in gillnets, 156 of those are expected to result in serious injury/mortality. Of the 213 loggerhead interactions expected to occur annually in the trawls, 62 of them are a result of unobservable, but quantifiable interactions in the summer flounder/scup/black sea bass fishery and are estimated to survive after escaping through TEDs in the nets. This leaves a balance of 151 observable interactions with loggerheads, 71 of those are expected to result in serious injury/mortality. As the serious injury/mortality rate for gillnets and trawls can also be applied to the other three sea turtle species, it is anticipated that the three of the four leatherback, two of the three Kemp's ridley, and three of the four green sea turtle interactions annually with gillnet gear, may result in serious injury/mortality. For bottom trawl gear, two of the four leatherback, two of the three Kemp's ridley, and two of the three green sea turtle interactions annually may result in serious injury/mortality.

<u>Serious injury/mortality calculations for trap/pot gear</u>
For black sea bass and scup trap/pot gear, the low occurrence of sea turtle interactions with this gear type does not allow for a valid determination of the anticipated level of lethal interactions. Therefore, the four annual interactions for leatherbacks and one annual interaction for loggerheads in the trap/pot fishery for scup/black sea bass could be either lethal or non-lethal.

### 7.6 Anticipated Interactions with Atlantic Sturgeon

The term take is defined under the ESA as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt or engage in any such conduct" and is typically described in terms of the impact to the individual fish – e.g. exposure to increased water temperature that results in injury, preventing access to spawning grounds, or capture in a fishing net. The life stage of the fish being impacted is also identified, as possible, when attributing take. In the case of Atlantic sturgeon, we have five separate DPSs, each of which is considered a separate species under the ESA, so we must attribute the fish taken to the appropriate DPS. A separate white paper has been prepared which provides the methodology that is used to make these assignments (see Damon-Randall *et al.* 2013).

The primary causes of sturgeon interactions from the seven fisheries are deployments of particular gear types in specific areas and time periods and attempts to quantify the degree of association between interactions and FMPs may be necessary for regulatory consideration, but the linkage between FMPs and sturgeon interactions is difficult to quantify. Attributing sturgeon interactions to individual FMPs is difficult because of the nature of fishing in the New England and Mid-Atlantic regions that results in species landed across multiple FMPs. The NEFSC conducted several analyses of sturgeon bycatch data and attempted to categorize interaction rates by commercially sought species groups (i.e. FMP species groups or proxies to FMP species groups). At the conclusion of their efforts, the NEFSC stated, "The partitioning of discard encounters to FMPs is not a particularly informative exercise because of the high likelihood of inappropriately attributing associations/ responsibilities." As noted previously, the pitfalls of partitioning Atlantic sturgeon bycatch by FMP was a major reason why we decided to batch these seven FMPs into this single consultation which allows us to identify, analyze, and address interactions of Atlantic sturgeon by gear type, area and time period.

The NEFSC Atlantic sturgeon bycatch report (2011b) analyzed fishery observer data and VTR data from fishermen in order to estimate the average annual number of Atlantic sturgeon interactions in gillnet and otter trawl gear in the Mid-Atlantic and New England regions that occurred in 2006-2010, the timeframe which included the most recent, complete data. This report on interactions represents the most accurate predictor of annual Atlantic sturgeon interactions in the fisheries.

The Atlantic sturgeon bycatch estimate methods for gillnets and otter trawls (NEFSC 2011) provide a quantitative association between the sturgeon encounters and gear types, as well as association to species groups within FMPs. Two processes were used to analyze the associations by gears and species groups: 1) a design based estimator (DBE) model was used to expand the ratio of total sturgeon takes to total landings by the total landings within a defined time and space (i.e. study cell); and, 2) a model based estimator (MBE) incorporated the mixture of species associated with the observed fishing trips which documented interaction with Atlantic sturgeon. The design based estimator relies on the assumption that discards are proportional to the total amount landed. While this has been observed for many species, the rarity of sturgeon makes it difficult to rely on this assumption. The MBE takes additional biological information into account and provides some information about the species associations that may influence sturgeon encounter rates. The model based approach allowed for a more comprehensive approach, therefore the results of the MBE are used throughout this Opinion.

### 7.6.1 Anticipated Interactions of Atlantic Sturgeon by Gear Type

The Atlantic sturgeon bycatch estimate methods for gillnets and trawls (NEFSC 2011) assigned trips 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 were estimated to have a certain bycatch rate of Atlantic sturgeon (based on the observed interactions). In the estimate, the gillnet trip and its associated Atlantic sturgeon interactions were assigned to several fisheries in a ratio that reflected the catch composition of that trip by weight. This method is meant to reflect the multispecies nature of many of the fisheries that operate throughout the Northeast and Mid-Atlantic regions.

Portions of the overall number of interactions reported in the NEFSC Atlantic sturgeon bycatch report (2011b) are attributed to striped bass and “other,” which includes state fisheries such as Atlantic croaker and non-targeted species such as lobster in gillnets. These should not be included in the interactions attributed to the FMPs included in this Opinion since they are not under the authority of the seven FMPs. Using the percentages in Table 32 and Table 33 we have excluded those interactions from the estimate of interactions attributed to the seven fisheries and instead considered those interactions as a component of the baseline and not a consequence of the proposed action. In Table 32 and Table 33, the base percentages for the “Sbass” and “other” categories, as well as the total interactions are as provided in the NEFSC bycatch report.

**Table 32**: Total otter trawl interactions with Atlantic sturgeon and proportion of take attributed to the seven fisheries as reported in the NEFSC bycatch report

| Year | Total Interactions | Proportion of take attributed to Sbass | Proportion of take attributed to Other | Total % not attributed to FMPs in batch (i.e. Sbass and Other) | Total # not attributed to FMPs in batch (i.e. Sbass and Other) | Total # attributed to FMPs in batch |
|---|---|---|---|---|---|---|
| 2006 | 1793.687 | 0.024 | 0.123 | 14.7% | 263.67 | 1530.02 |
| 2007 | 1645.893 | 0.02 | 0.121 | 14.1% | 232.07 | 1413.82 |
| 2008 | 1392.025 | 0.013 | 0.114 | 12.7% | 176.79 | 1215.24 |
| 2009 | 1338.139 | 0.013 | 0.122 | 13.5% | 180.65 | 1157.49 |
| 2010 | 1570.297 | 0.007 | 0.109 | 11.6% | 182.15 | 1388.14 |
| average | 1548.008 | 0.0154 | 0.118 | 13.3% | 206.19 | 1341.81 |

**Table 33**: Total gillnet interactions with Atlantic sturgeon and proportion of take attributed to several fisheries, as reported in the NEFSC bycatch report

| Year | Total Interactions | Proportion of take attributed to Sbass | Proportion of take attributed to Other | Total % not attributed to FMPs in batch (i.e. Sbass and Other) | Total # not attributed to FMPs in batch (i.e. Sbass and Other) | Total # attributed to FMPs in batch |
|---|---|---|---|---|---|---|
| 2006 | 1612.001 | 0.043 | 0.23 | 27.3% | 440.08 | 1171.92 |
| 2007 | 2216.112 | 0.107 | 0.115 | 22.2% | 491.98 | 1724.14 |
| 2008 | 858.155 | 0.092 | 0.108 | 20.0% | 171.63 | 686.52 |
| 2009 | 2053.346 | 0.045 | 0.176 | 22.1% | 453.79 | 1599.56 |
| 2010 | 1107.961 | 0.008 | 0.13 | 13.8% | 152.90 | 955.06 |
| average | 1569.515 | 0.059 | 0.1518 | 21.1% | 330.85 | 1238.66 |

**Table 34**: Interactions of the seven fisheries with Atlantic sturgeon by gear types: otter trawls and sink gillnets

| Year | Total Interactions | Otter Trawl | | Gillnet | |
|---|---|---|---|---|---|
| | | # | % | # | % |
| 2006 | 2701.94 | 1530.02 | 56.63% | 1171.92 | 43.37% |
| 2007 | 3137.96 | 1413.82 | 45.06% | 1724.14 | 54.94% |
| 2008 | 1901.76 | 1215.24 | 63.90% | 686.52 | 36.10% |
| 2009 | 2757.05 | 1157.49 | 41.98% | 1599.56 | 58.02% |
| 2010 | 2343.20 | 1388.14 | 59.24% | 955.06 | 40.76% |
| average | 2580.47 | 1341.81 | 52.00% | 1238.66 | 48.00% |

### *Otter Trawls*

As shown in Table 32 above, based on data collected by observers for reported Atlantic sturgeon captures in bottom otter trawl gear, the NEFSC estimated the average annual bycatch of Atlantic sturgeon in bottom otter trawl gear during 2006-2010 to be 1,341.81 (NEFSC 2011). This estimate of Atlantic sturgeon bycatch in bottom otter trawl gear provides the best available information for determining the anticipated number of Atlantic sturgeon interactions per year in the bottom trawl components of the seven fisheries. For the purposes of this Opinion, we are rounding the annual average of 1,341.81 to 1,342 since a partial sturgeon take is not possible. Thus, up to **1,342** Atlantic sturgeon per year is the best available information on the anticipated number of interactions in the bottom trawl component of these fisheries. This represents the total number of interactions we are expecting annually in the bottom trawl component of these fisheries and not just the number observed.

***Gillnets***

As shown in Table 34 above, from 2006 to 2010, the average annual bycatch estimate of Atlantic sturgeon in Northeast and Mid-Atlantic gillnet gear was 1,238.66 individuals (NEFSC 2011). For the purposes of this Opinion, we are rounding the annual average of 1,238.66 to **1,239** since a partial sturgeon take is not possible. These estimates of Atlantic sturgeon interactions with Northeast and Mid-Atlantic gillnet gear provide the best available information for determining the anticipated bycatch of Atlantic sturgeon in that gear type in the action area. This represents the total number of interactions we are expecting annually in the gillnet component of these fisheries and not just the number observed.

### 7.6.2 Estimated Mortalities and Age Classes of Atlantic Sturgeon that Interact with Gear Types

NEFOP data indicates that mortality rates of Atlantic sturgeon caught in otter trawl gear and gillnet gear is approximately 5% and 20%, respectively. NEFOP data also indicates that of the Atlantic sturgeon interactions that have been observed, approximately 75% are subadults and 25% are adults based on length (n=726; subadults less than 150cm, adults 150cm or longer). More specifically, the encountered ratios for gillnet gear were approximately 72% subadults to 28% adults and the ratios for trawl gear were 79% subadults to 21% adults.

Damon-Randall *et al.* (2013) used NEFOP information regarding Atlantic sturgeon interactions in conjunction with genetic testing results of Atlantic sturgeon sampled through the NEFOP to calculate percentages of each DPS represented in the Northeast region: Gulf of Maine DPS at 11%; New York Bight DPS at 51%; Chesapeake Bay DPS at 13%; Carolina DPS at 2%; South Atlantic DPS at 22%; and Canada at 1% (*i.e.*, from the St. John River). Since data were lacking to calculate total population estimates, we used the NEAMAP-based estimates for ocean populations and the mixed stock analysis genetics results presented in Damon-Randall *et al.* 2013 for each DPS. Next, we were able to calculate an “adult equivalent” rate which converts a number of subadults to adults (the number of subadults that would, through natural mortality, live to be adults). This was calculated by dividing the total number of fish in any one year aged 11-20 (i.e. adults) by the total number of fish aged 2-10 (i.e. subadults) to determinethe number of adults per sub-adult. When using the age-variable mortality rate (M) given in Kahnle *et al.* (2007), the result is 0.48. No estimate was given in Kahnle *et* *al.* (2007) for the M for age 1 sturgeon, so we assumed it was the same as for age 2, which was 0.16. We then converted numbers of subadults estimated to be affected by the proposed action to adults. The effects analysis for this Opinion will necessarily consider impacts to subadults in addition to the adult take estimate (which includes adults and adult equivalents). We do not have information at this time to complete this type of "adult equivalent" calculation for other life stages (i.e., early life stages such aseggs or larvae, young of the year, or juveniles); however, that is unnecessary for this consultation since only subadult and adult life stages are likely to be impacted by the proposed action.

**Table 35** Estimated mortalities by DPS for the batched FMPs based on NEFOP data 2006-2010. DPS percentages listed are the point values representing the genetics mixed stock analysis results.

**Batched FMPs**

**Sink Gillnet**

| | | | Dead Encounters by Life Stage | | Dead Encounters: Adults Plus Adult Equivalents |
|---|---|---|---|---|---|
| | % Mortality | Estimated Dead Encounters | 27.67% adult | 72.33% subadult | |
| Avg (1238.66) | 0.20 | 247.73 | 68.55 | 179.18 | 154.56 |
| GOM (11%) | | 27.25 | 7.54 | 19.71 | 17.00 |
| NYB (51%) | | 126.34 | 34.96 | 91.38 | 78.82 |
| CB (13%) | | 32.21 | 8.94 | 23.30 | 20.12 |
| Carolina (2%) | | 4.96 | 1.37 | 3.59 | 3.09 |
| SA (22%) | | 54.5 | 14.92 | 39.42 | 33.84 |
| Canada (1%) | | 2.48 | 0.69 | 1.79 | 1.55 |

**Otter Trawl**

| | | | Dead Encounters by Life Stage | | Dead Encounters: Adults Plus Adult Equivalents |
|---|---|---|---|---|---|
| | % Mortality | Estimated Dead Encounters | 20.54% adult | 79.46% subadult | |
| Avg (1341.81) | 0.05 | 67.09 | 13.78 | 53.31 | 39.37 |
| GOM (11%) | | 7.38 | 1.52 | 5.86 | 4.33 |
| NYB (51%) | | 34.22 | 7.03 | 27.19 | 20.08 |
| CB (13%) | | 8.72 | 1.79 | 6.93 | 5.12 |
| Carolina (2%) | | 1.34 | 0.28 | 1.07 | 0.79 |
| SA (22%) | | 14.76 | 3.03 | 11.73 | 8.66 |
| Canada (1%) | | 0.67 | 0.14 | 0.53 | 0.39 |

**Total - Sink Gillnet and Otter Trawl**

| | Estimated Dead Encounters | Dead Encounters by Life Stage: Adult | Dead Encounters by Life Stage: Subadult | Dead Encounters: Adults Plus Adult Equivalents |
|---|---|---|---|---|
| **avg** | 314.82 | 82.33 | 232.49 | 193.93 |
| **GOM (11%)** | 34.63 | 9.06 | 25.57 | 21.33 |
| **NYB (51%)** | 160.56 | 41.99 | 118.57 | 98.90 |
| **CB (13%)** | 40.93 | 10.73 | 30.23 | 25.24 |
| **Carolina (2%)** | 6.30 | 1.65 | 24.37 | 13.35 |
| **SA (22%)** | 69.26 | 17.95 | 51.15 | 42.50 |
| **Canada (1%)** | 3.15 | 0.83 | 2.32 | 1.94 |

## 7.7 Anticipated Interactions with Atlantic Salmon

Due to the low number of observed interactions and the low number of Atlantic salmon in the action area, it is expected that interactions between the seven fisheries and Atlantic salmon will be low, and possibly non-existent, in any given year.

### 7.7.1 Anticipated Interactions by Gear Type

There are no bycatch estimates for Atlantic salmon in gillnet or trawl gear. The very low number of observed Atlantic salmon interactions in gillnet and trawl gear as reported in the NEFOP database (which includes ASM data) suggests that interactions within the action area are rare events. However, given the fact that observer coverage in these fisheries is less than 100%, it is likely that some interactions with Atlantic salmon have occurred but were not observed or reported. Due to the effort in the fisheries as a whole, and the seasonal overlap in distribution of these species with operation of gillnet and trawl gear, a small number of Atlantic salmon may interact with both gear types.

### 7.7.2 Gillnets

A review of the NEFOP and ASM observer records from 1989 through 2011 reveals that 9 of 13 (69%) incidental takes occurred in sink gillnet gear. The average annual number of Atlantic salmon captures in gillnet gear in the action area documented through the NEFOP and ASM is 0.41 (9 divided by 22). Since the capture of a partial Atlantic salmon is not possible, we round that number to one per year.

### 7.7.3 Bottom Trawls

A review of the NEFOP and ASM observer records from 1989 through 2011 reveals that 4 of 13 (31%) incidental takes occurred in bottom trawl gear. The average annual number of Atlantic salmon captures in botton trawl gear in the action area documented through the NEFOP and ASM is 0.18 (4 divided by 22).

Since the capture of a partial Atlantic salmon is not possible, we round 0.18 to one per year.

#### 7.7.4 Estimated Mortality

Of the 13 total reported interactions with Atlantic salmon in the seven fisheries, at least five resulted in mortalities. Eight are listed as "discarded" in the database, and are assumed to have been discarded alive. The seven fisheries assessed in this Opinion primarily use sink gillnet and bottom otter trawl gear. A review of the observer records from 1989 through 2011 reveals that 9 of 13 (69%) incidental takes occurred in sink gillnet gear, with the remaining 4 of 13 (31%) occurred in bottom otter trawl gear. Of the nine incidental takes in sink gillnet gear, three were dead (33%), while six were discarded presumed alive (66%). Of the four incidental takes in bottom otter trawl gear, two were dead (50%) and two were discarded presumed alive (50%). Thus, one-third of the interactions in sink gillnet gear are expected to result in mortalities, and half of the interactions with bottom otter trawl gear are expected to result in mortalities. It is anticipated that an annual average of up to one Atlantic salmon take in gillnet gear may occur annually in the seven fisheries, with one lethal take occurring on average every three years. Additionally, it is anticipated that an annual average of up to one Atlantic salmon take in bottom trawl gear may occur annually in the seven fisheries, with one lethal take occurring on average every two years.

### 7.8 Summary of Anticipated Interactions with ESA-listed Species

#### 7.8.1 Whales

The primary gear types used in the seven fisheries are bottom trawls, sink gillnets, and hook and line gear. Although large whale entanglements in trawl and hook gear has been documented, these are rare events relative to gillnet entanglements, and are not expected to result in SI/M. Based on results from large whale entanglements analyses, NMFS believes the greatest risk to whales from the seven fisheries is entanglements in gillnet gear.

Based on NMFS' large whale entanglement data for the years 2006-2010, the annual mean rates of fin whale and sei whale entanglements resulting in serious injury or mortality (SI/M) have been 0.8 and 0.6, respectively. The type of gear was unidentified in 100% of the fin and sei whale entanglement events. We anticipate zero to three and zero to two annual entanglements resulting in SI/M being detected for fin and sei whales respectively.

The 2012 SAR has the annual mean rate of SI/M from fishery gear entanglements listed as 1.6 and 5.2, respectively, for right and humpback whales in U.S. waters for 2006-2010 (Waring *et al.* 2012). During that period, one of the entangled humpbacks in sink gillnet gear resulted in a mortality and one of the entangled right whales in unspecified gillnet gear resulted in a serious injury. The seven fisheries do pose a risk of serious injury and mortality to right and humpback whales as a result of entanglement in gillnet gear. We anticipate the range of detected entanglements resulting in SI/M as a result of U.S. fishing gear to be zero to three for North Atlantic right whales and zero to eight for humpback whales (NMFS NERO 2012). The continued implementation and development of ALWTRP measures, along with an overall reduction in fishery effort provide cause to anticipate the number of right and humpback whale entanglements in gillnet gear should decline or, at least, not increase.

### 7.8.2 Sea Turtles

Based on information from Murray (2009a), Warden (2011a), and the STDN, we anticipate up to 483 loggerhead sea turtles from the NWA DPS will interact annually with gear utilized in the seven fisheries assessed in this Opinion. An average of up to 269 loggerheads are expected to interact with gillnet gear annually based on the upper ends of the 95% CIs for the bycatch estimates by FMP group in Murray (2009a). In addition, an average of up to 213 loggerheads are expected to interact annually with bottom trawl gear, based on the upper ends of the 95% CIs for the bycatch estimates by FMP group in Warden (2011a). Also, up to one loggerhead is expected to interact annually with trap/pot gear in the black sea bass/scup fishery. Fifty-eight percent (156) of the annual interactions in gillnet gear and 47% (71) of the observable annual interactions in bottom trawl gear are expected to lead to serious injury or mortality, while the one loggerhead interaction in trap/pot gear could possibly be lethal. Therefore, up to 228 of the 483 loggerhead sea turtles that interact with these fisheries annually are expected to die or sustain serious injuries leading to death or failure to reproduce.

Based on fishery observer data from the NEFOP and ASM programs, we anticipate up to four leatherback sea turtle interactions annually (up to three lethal) with gillnet gear and up to four interactions annually (up to two lethal) with bottom trawl gear used in these fisheries. Based on data from the STDN, we also expect up to four annual leatherback interactions with black sea bass/scup trap/pot gear, which could be lethal or non-lethal. For Kemp's ridley sea turtles, we anticipate up to three interactions with gillnet gear (up to two lethal) and up to three interactions with bottom trawl gear (up to two lethal) will occur annually as a result of these fisheries. Finally, for green sea turtles, we anticipate up to four interactions with gillnet gear (up to three lethal) and up to three interactions with bottom trawl gear (up to two lethal) will occur annually as a result of these fisheries. The anticipated annual interaction rates for Kemp's ridley and green sea turtles are based on observer data from both the NEFOP and ASM programs.

A summary of the annual anticipated sea turtle interactions in the seven fisheries addressed in this Opinion is summarized by gear type below:

**Table 36** Anticipated sea turtle interactions (mortalities) by gear type in the batched fisheries

| | Gillnet Interactions (Mortalities) | Trawls Interactions (Mortalities) | Trap/Pot Interactions (Mortalities) | Total Interactions (Mortalities) |
|---|---|---|---|---|
| Loggerheads | 269 (156) | 213(71) | 1 (0-1) | 483 (up to 228) |
| Leatherbacks | 4 (3) | 4 (2) | 4 (0-4) | 12 (up to 9) |
| Kemp's Ridleys | 3 (2) | 3 (2) | | 6 (4) |
| Greens | 4 (3) | 3 (2) | | 7 (5) |

### 7.8.3 Atlantic Sturgeon

Based on the history of documented interactions with commercial fishing gear and largely on the results of the NEFSC Atlantic sturgeon bycatch report (2011b) which analyzed NEFOP data and VTR data from fishermen, we anticipate up to 2581 interactions annually between Atlantic sturgeon and otter trawls and gillnets used in the batched fisheries. Of those interactions, 1342 are expected to be with otter trawls and 1239 are expected from gillnet gear.

NEFOP data indicates that average mortality rates of Atlantic sturgeon caught in otter trawl gear and gillnet gear across the federal fisheries is approximately 5% and 20%, respectively. Additionally, NEFOP data indicates that of the Atlantic sturgeon interactions that have been observed, the encountered ratios for gillnet gear were approximately 72% subadults to 28% adults and the ratios for trawl gear were 79% subadults and 21% adults. Using those percentages and results from the genetics mixed stock analysis we have been able to estimate the number of subadults and adult interactions and mortalities with each gear type per DPS.

Next, we were able to calculate an "adult equivalent" rate which converts a number of subadults to adults (the number of subadults that would, through natural mortality, live to be adults). We then converted numbers of subadults estimated to be affected by the proposed action to adults through an adult equivalent calculation which allowed us to consider impacts to subadults in addition to the adult take estimate.

A summary of the annual anticipated Atlantic sturgeon interactions in the seven fisheries addressed in this Opinion is summarized by gear type below:

**Table 37** Anticipated Atlantic sturgeon adults plus adult equivalents interactions (mortalities) by gear type in the batched fisheries

| | Gillnet Interactions of Adults and Subadults | Gillnet Mortalities of Adults and Subadults (adults + adult equivalents) | Trawls Interactions of Adults and Subadults | Trawl Mortalities of Adults and Subadults (adults + adult equivalents) |
|---|---|---|---|---|
| GOM DPS | 137 | 28 (17) | 148 | 8 (5) |
| NYB DPS | 632 | 127 (79) | 685 | 35 (21) |
| CB DPS | 162 | 33 (21) | 175 | 9 (6) |
| Carolina DPS | 25 | 5 (4) | 27 | 2 (1) |
| SA DPS | 273 | 55 (34) | 296 | 15 (9) |
| Canada | 13 | 3 (2) | 14 | 1 (1) |

### 7.8.4 Atlantic Salmon

Historical data from 1989 through 2011 shows that there have been 13 observed interactions between Atlantic salmon and otter trawls and gillnets used in the seven fisheries. Given that post-smolt Atlantic salmon rapidly migrate through the Gulf of Maine and all captured salmon weighed at least one pound, we assume that they are all subadults (Baum 1997, Lacroix *et al.* 2012). Lacking genetic information of the fish involved in these interactions and based on the known distribution of GOM DPS Atlantic salmon, taking a precautionary approach, we are assuming the interactions were GOM DPS Atlantic salmon. Based on past data, we anticipate two GOM DPS Atlantic salmon interactions on average annually, with one of the interactions involving gillnet gear and one in bottom trawl gear. A lethal take is expected to occur on average every three years in gillnet gear and on average every two years in bottom trawl gear.

## 8.0 Cumulative Effects

Cumulative effects as defined in 50 CFR 402.02 include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area considered in this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA. For that reason, future effects of other federal fisheries are not considered in this section of the document; all federal fisheries that may affect listed species are the subject of formal section 7 consultations. Effects of ongoing federal activities, including other fisheries, are considered in the Environmental Baseline and Status of the Species sections of this Opinion and are also factored into the Integration and Synthesis of Effects section below.

Sources of human-induced mortality, injury, and/or harassment of marine mammals, sea turtles, Atlantic sturgeon, and Atlantic salmon in the action area that are reasonably certain to occur in the future include interactions in state-regulated and recreational fishing activities, vessel collisions, ingestion of plastic debris, pollution, global climate change, coastal development, and catastrophic events. While the combination of these activities may affect populations of ESA-listed marine mammals, sea turtles, Atlantic sturgeon, and Atlantic salmon preventing or slowing a species' recovery, the magnitude of these effects is currently unknown.

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

Fishing activities are considered one of the most significant causes of death and serious injury for sea turtles. Fishing gear in state waters, including bottom trawls, gillnets, trap/pot gear, and pound nets, interacts with sea turtles each year. NMFS is working with state agencies to address interactions of sea turtles in state-water fisheries within the action area of this consultation where information exists to show that these fisheries interact with sea turtles. Action has been taken by some states to reduce or remove the likelihood of sea turtle interactions in one or more gear types. However, given that state managed commercial and recreational fisheries along the U.S. Atlantic coast are reasonably certain to occur within the action area in the foreseeable future, interactions of sea turtles with these fisheries are anticipated. There is insufficient information on the number of sea turtle interactions presently occurring in state water fisheries and on the number of sea turtles injured or killed as a result. While actions have been taken to reduce sea turtle interactions in some state water fisheries, the overall effect of these actions is unknown, and the future effects of state water fisheries on sea turtles cannot be quantified.

Right and humpback whale entanglements occur in gear set in state waters. Entanglements in state lobster pot/traps and in croaker sink gillnet gear have been reported (Waring *et al.* 2007; Glass *et al.* 2008). Actions have been taken to reduce the risk of entanglement to large whales, although more information is needed to assess 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* – 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, although it is not always obvious whether these injuries were pre- or post-mortem (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 or kill sea turtles, and many stranded turtles have obvious propeller or collision marks (Dwyer *et al.* 2003). However, it is not always clear whether the collision occurred pre- or post-mortem. NMFS believes that vessel interactions with sea turtles will continue. An estimate of the number of sea turtles that will likely be killed by vessels is not available from data at this time.

This Opinion assumes effects in the future would be similar to those in the past and are, therefore, reflected in the anticipated trends described in the *Status of the Species* and *Environmental Baseline* sections. As indicated above, vessel interactions do not appear to be a threat to Atlantic salmon

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 the *Environmental Baseline* section of this document, NMFS has implemented a ship strike reduction program to reduce the number of right whale strikes by large vessels. The program consists of both regulatory and non-regulatory components, such as requiring vessels to reduce speed in certain areas at certain times when right whales are likely to be present. The program is not specific to areas or times when other species of large whales are likely to be present in the vicinity of large ports of shipping lanes. The program does not require reduced speeds in all areas where right whales may occur. Although these measures are designed to reduce interactions of ESA-listed whales as a result of vessel strikes, the risk of interaction has not been fully removed since interactions may still occur at times when large whales and vessels occupy the same areas.

*Pollution and Contaminants* – Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from pollution on ESA-listed marine mammals, sea turtles, and fish. However, the level of impacts cannot be projected. Sources of contamination in the action area include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, and industrial development. Chemical contamination may have effects on listed species' reproduction and survival. Excessive turbidity due to coastal development and/or construction sites could influence marine mammal, sea turtle, or fish foraging ability. Marine debris (*e.g.*, discarded fishing line or lines from boats) also has the potential to entangle marine mammals and sea turtles in the water or to be fed upon by them. Sea turtles commonly ingest plastic or mistake debris for food and sometimes this may lead to asphyxiation. This Opinion assumes effects in the future would be similar to those in the past and are therefore reflected in the anticipated trends described in the *Status of the Species* and *Environmental Baseline* sections.

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, which raises new concerns about their effects on right whales (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008). The impacts of biotoxins on marine mammals are also poorly understood, yet data is showing that marine algal toxins may play significant roles in mass mortalities of these animals (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers have discovered that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer through the copepods upon which they feed (Durbin *et al.* 2002; Rolland *et al.* 2007; Leandro *et al.* 2009).

Other large whales are likely similarly affected. Between November 1987 and January 1988, at least 14 humpback whales died after consuming Atlantic mackerel containing a dinoflagellate saxitoxin (Geraci *et al.* 1989; Waring *et al.* 2009). In July 2003, dead humpback whales tested positive for low levels of domoic acid (Waring *et al.* 2009). However, domoic acid poisoning could not be confirmed as the cause of death (Waring *et al.* 2009).

Noise pollution has been raised primarily as a concern for marine mammals but may be a concern for other marine organisms, including sea turtles. The potential effects of noise pollution on marine mammals and sea turtles range from minor behavioral disturbance to injury and death. The noise level in the ocean is thought to be increasing at a substantial rate due to increases in shipping, seismic exploration, offshore drilling, and sonar used by military and research vessels (NMFS 2007b). Because under some conditions low frequency sound travels very well through water, few oceans are free of 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 sound by another, could interfere with marine mammals' ability to feed and to communicate for mating (NMFS 2007b). Masking is a major concern about shipping, but only a few species of marine mammals have been observed to demonstrate behavioral changes to low level sounds. Concerns about noise in the action area are primarily related to increasing commercial shipping and recreational vessels.

*Global Climate Change* - In the future, global climate change is expected to continue and may impact ESA-listed marine mammals, sea turtles, fish, and their habitats in the action area. However, as noted in the *Status of the Species* and *Environmental Baseline* sections above, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate related impacts will have a significant effect on the status of any species of marine mammals, sea turtles, or fish in the short-term future (*i.e.*, over the next decade or so) or that in this time period, the abundance, distribution, or behavior of these species in the action area will change as a result of climate change related impacts.

*Coastal Development* – Along the Mid-Atlantic and Southeast coastline, beachfront development, lighting, and beach erosion 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. 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 negative effects to hatchlings.

*Hydroelectric Dams* – Hydroelectric facilities can alter the river's natural flow pattern and temperatures, affecting Atlantic salmon and Atlantic sturgeon. In addition, the release of silt and other fine river sediments during dam maintenance can be deposited in sensitive spawning habitat nearby. These facilities also act as barriers to normal upstream and downstream movements, and block access to important habitats. Passage through these facilities may result in the mortality of upstream and downstream migrants.

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

## 9.0 Integration and Synthesis of Effects

The *Status of Affected Species, Environmental Baseline,* and *Cumulative Effects* sections of this Opinion discuss the natural and human-related phenomena that caused right, humpback, fin and sei whales; loggerhead, leatherback, Kemp's ridley and green sea turtles; the five DPSs of Atlantic sturgeon; and GOM DPS Atlantic salmon to become endangered or threatened and may continue to place the species at high risk of extinction. "Jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR 402.02). The present section of this Opinion applies that definition by examining the effects of the proposed action in the context of information presented in the status of the species, environmental baseline, and cumulative effects sections to determine: (a) if the effects of the proposed action would be expected to reduce the reproduction, numbers, or distribution of the previously listed cetaceans, sea turtles, and fish, and (b) if any reduction in the reproduction, numbers, or distribution of these species causes an appreciable reduction in the species' likelihood of surviving and recovering in the wild.

In the NMFS/U.S. Fish and Wildlife Section 7 Handbook, "survival" is defined as:

> 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.

This Opinion has identified in section 7.0 (*Effects of the Proposed Action*) that the proposed action—continued operation of the seven fisheries—may directly affect right, humpback, fin, and sei whales as a result of entanglement in gear fished in the seven fisheries. No other direct or indirect effects to ESA-listed cetaceans are expected as a result of the activity. This Opinion has also identified that the proposed action may directly affect loggerhead, leatherback, Kemp's ridley and green sea turtles, as well as Atlantic sturgeon and Atlantic salmon, as a result of interaction with gear used in the seven fisheries. No other direct or indirect effects to ESA-listed sea turtles, Atlantic sturgeon or Atlantic salmon are expected as a result of this activity. The discussion below provides NMFS' determinations of whether there is a reasonable expectation that right, humpback, fin, and sei whales; loggerhead, leatherback, Kemp's ridley, and green sea turtles; Atlantic sturgeon; and Atlantic salmon will experience reductions in reproduction, numbers, or distribution in response to these effects, and whether any reductions in the reproduction, numbers, or distribution of these species can be expected to appreciably reduce the species' likelihood of surviving and recovering in the wild. It is important to consider that the assessments in sections 9.1 through 9.9 are based on historical data and do not fully account for the trend in reduction of effort in the seven fisheries and other fisheries. Thus, the assessments in these sections could be considered worst case expectations as the relatively recent reductions in commercial fisheries effort could result in decreased opportunities for interactions of ESA-listed species.

## 9.1 North Atlantic Right Whale

As described in the Status of Species section of this Opinion, for 2006-2010, the average reported mortality and serious injury to right whales due to fishery entanglement was 1.8 whales per year (U.S. waters, 1.6; Canadian waters, 0.2) (Waring *et al.* 2012). In the majority of cases, an entanglement report does not contain the necessary information to assign the event to a particular fishery. From 2006-2010, gillnet gear of U.S. or undocumented origin was recorded in seven entanglement events with right and humpback whales (Table 24). Of those seven events, unknown gillnet gear was verified to be involved with the entanglement of one right whale and sink gillnet gear was not verified to be involved with any entanglement of right whales. Although there are no documented cases of SI/M to right whales from sink gillnet gear in 2006-2010, SI/M has previously been documented for right whales as a result of entanglement in sink gillnet gear. Based on the serious injury and mortality data for the past 10 years, we expect to document a range of zero to three right whales seriously injured or killed per year as a result of entanglement in U.S. fishing gear.

For the purposes of this assessment, we are assuming that on a five-year average, zero to three right whales are documented as seriously injured or killed as a result of U.S. fisheries. Under the worst case scenario, we could have five years in a row where three serious injuries or mortalities were reported, resulting in an average of three per year. Therefore, we expect the five-year average to range from zero to three. Because serious injury or mortality could result from the seven fisheries, this Opinion assumes that serious injury or mortality could and would occur as a result of the seven fisheries.

PBR for the western Atlantic stock of North Atlantic right whale stock is 0.9 whales (Waring *et al.* 2012). As indicated above, while the annual average rate of documented SI/M events for right whales attributable to gillnet gear is less than PBR (0 < 0.2), the overall annual rate of documented serious injury/mortality events with all U.S. commercial fishing gear for right whales is 1.6, which exceeds the PBR value of 0.9. 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 the population size necessary for survival and recovery. The 2012 SAR indicates that the level of serious injuries or mortalities of North Atlantic right whales attributable to U.S. commercial fisheries exceeds the level necessary to allow for growth to the optimum sustainable population level. However, what we must consider in this Opinion is whether the continued operation of the seven fisheries over the next ten years will result in interactions with right whales that will result in serious injuries or mortalities that are likely to appreciably reduce the survival and recovery of North Atlantic right whales. If so, then we would have to determine if that appreciable reduction in survival and recovery for the western Atlantic stock resulted in an appreciable reduction in survival and recovery for North Atlantic right whales.

As described in the Status of Species section of this Opinion, the latest final stock assessment report indicates that the population of North Atlantic right whales has grown at a rate of 2.6% between 1990 and 2009 (Waring *et al.* 2012), so while SI/M have exceeded PBR, the population is still increasing. 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, unpublished). 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 Appendix A to this Opinion, but some of the relevant results are 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 (over a 100-year period).
- Only 2 of 1000 projections (with status quo simulation over a 100-year period) ended the 100 year period with a smaller total population size than they started with (345), 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%; with one less adult female mortality per year, the probability improved to 24.6%.

*Effects of Serious Injury or Mortality from Fisheries Entanglement on Survival and Recovery*

The modeling done by Pace (unpublished) indicates that under the status quo (i.e., no changes in mortality rate) there is a very low likelihood of the North Atlantic right whale going extinct or reaching a quasi-extinction level (a population of only 50 adult females, see explanation below). None of the model projections actually predicted extinction or quasi-extinction. Agreed upon criteria for quasi-extinction, *i.e*., population numbers, structure and trends, for North Atlantic right whales have not yet been developed; however, quasi-extinction 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 PVA conducted by Pace (unpublished) used a quasi-extinction level of 50 adult female right whales. The rationale for this level follows: (1) there is general consensus in the conservation genetics community that large vertebrate populations cannot fall below 50 breeding animals and still maintain genetic integrity (Shaffer 1981;

Franklin 1980), and (2) the International Union for Conservation of Nature (IUCN)(Reilly *et al.* 2008) considers this to be one of the two threshold numerical values for a "critically endangered" population category (IUCN 2008). IUCN uses 250 mature animals as an alternative threshold value for "critically endangered" populations when there is evidence of a population decline. Given the population increase currently observed for the species (2.6% increase from 1990-2009 (Waring *et al.* 2012), or 1.3% (Pace, unpublished) based on the parameters and time series in his model), it is reasonable to use 50 rather than 250 as the threshold value for quasi-extinction. As described above, using 50 adult females as the quasi-extinction threshold, Pace (unpublished) observed zero simulations out of 1,000 getting to 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 population to successfully reproduce.

While the mortality of zero to three right whales per year will reduce the number of right whales in the population compared to the number that would have been present absent the proposed action, as evidenced by the results of the PVA, it is not likely that this reduction in numbers will appreciably reduce the likelihood of survival and recovery. As described above, none of the 1,000 runs of the status quo projections in the PVA, which assumes future levels of serious injury and mortality due to U.S. fishing gear are similar to past levels, predict extinction. In addition, only two of the 1,000 status quo projections ended the 100 year period with a smaller total population size than the starting population size.

Reproductive potential of North Atlantic right whales is not expected to be affected in any other way other than through a reduction in numbers of individuals. The mortality of zero to three right whales per year would have the effect of reducing the amount of potential reproduction of right whales as the right whales killed would have no potential for future reproduction. However, future reproductive value was considered in the PVA, and, as evidenced by the results of the PVA, a reduction in the current mortality level by one animal per year, even a mature female, does not change the future trajectory of this species. Even considering the potential loss of future mature whales that would be produced by the individuals that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be very small and would not change the increasing trend of this population. Additionally, the proposed action will not affect habitat in any way that will reduce mating or rearing success.

The proposed action is not likely to reduce distribution because the action will not prevent right whales from accessing any habitats used seasonally for migrating, foraging, mating or rearing.

While generally speaking, the loss of a small number of individuals from a subpopulation or species can have an appreciable effect 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. The results of the PVA indicate that this is not the case for right whales and the loss of individuals as a result of entanglement in fishing gear, at a rate similar to what has occurred in the past, is not likely to appreciably reduce the likelihood of survival of this species (*i.e.*, it will not appreciably increase the risk of extinction faced by this species).

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that North Atlantic right whales will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., "endangered"), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., "threatened") because of any of the following five listing factors: (1) the present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The proposed action is not expected to modify, curtail, or destroy the range of the species since it will result in the annual mortality of zero to three individuals and the PVA indicates that this loss will not cause an appreciable change in the increasing trend of this population and therefore it will not affect the overall distribution of right whales. The proposed action will not utilize right whales for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The loss of these individuals will not change the status or trend of the species, which is increasing, and would not result in an appreciable reduction in the likelihood of improvement in the status of right whales throughout their range. The effects of the proposed action will not hasten the extinction timeline or otherwise increase the danger of extinction. Below, we consider effects of the action on the downlisting criteria identified for right whales in the most recent recovery plan.

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

The revised Recovery Plan for North Atlantic Right Whales states that the most significant need for North Atlantic right whale recovery is to reduce or eliminate deaths and injuries from anthropogenic activities, namely shipping and commercial fishing operations. As described in this Opinion, there are numerous management and regulatory initiatives implemented and underway to meet this need. Several significant management measures have been implemented recently, and their effects would not yet be expected to be seen in the population in terms of an increased population growth rate. Two of the more significant measures designed to reduce the risk from these anthropogenic activities are the implementation of the ALWTRP measures in 2009 (*e.g.*, broad based gear modifications requiring the use of sinking groundlines for gillnet and pot/trap gear) and the Ship Strike Reduction Program, including the 2008 regulations requiring large ships to reduce speeds to 10 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 (unpublished), nor are they included in the latest stock assessment report (Waring *et al.* 2012). 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 2009a). 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 (unpublished), 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 (unpublished), 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 (2.6% for the 1990-2009 period), however, the population is increasing at a rate targeted for downlisting (if maintained for 35 years) as identified in the species' recovery plan. It is important to note that the median growth rates (including under the status quo) in Pace (unpublished) are based on model simulations, while the population growth rate of 2.6% in Waring *et al.* (2012) is an observed growth rate in the population. The modeling uses a longer timeframe that 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 2.6% over a 19-year period (1990–2009) indicates that if the status quo continues and this growth rate is maintained, the downlisting criteria will be met. The population appears to be on the correct trajectory to meet the downlisting criteria if the status quo can be maintained.

An additional downlisting criteria states that the right whale population should have no more than a 1% chance of quasi-extinction in 100 years. As stated previously, none of the 1,000 runs of the PVA status quo projections resulted in a prediction of quasi-extinction in 100 years. Therefore, the population currently appears to be meeting this downlisting criteria.

Based on this analysis, the effects of the proposed action will not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that right whales can be brought to the point at which they are no longer listed as endangered or threatened.

Another important factor to consider, as noted above, 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 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, 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 by NMFS to its implementation within a given time schedule (as described in Section 4.4.5.1). Additionally, fishing effort in the seven fisheries has been reduced.

As described above and as indicated in Pace (unpublished), 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 targeted for downlisting (if maintained for 35 years) as identified in the species' recovery plan. The species has persisted and is projected to do so into the future. The projected and observed mean population growth for the past 19 years provides evidence that the species has sufficient resilience to allow for recovery from endangerment. It is important to consider that the action being considered in this Opinion is not new, it is ongoing and the right whale population has been increasing while the seven fisheries have continued to occur and continued to impact right whales. No changes to the fishery are being proposed that would increase the potential for interactions between the fishery and right whales.

Based on the analysis described above, the serious injury or mortality of zero to three right whales per year as a result of fisheries entanglement in U.S. gear over the next ten years is not likely to reduce appreciably the likelihood of both survival and recovery of North Atlantic right whales.

### 9.2 Humpback Whale

As established above, the use of gillnet gear for the proposed activity is expected to result in the entanglement of humpback whales. An annual average of 4.4 SI/M events of humpbacks in fishing gear has been documented for the period 2006-2010 (NMFS NERO 2012). During that same time period, the average documented SI/M events for humpbacks in gillnet entangling gear was 0.2 (NMFS NERO 2012). It should be noted that this database includes a large number of entanglements with undocumented gear types, which may include non-fishery related gear like anchoring systems and mooring gear. Another accounting of serious injury/mortality events for humpback whales from 2006-2010 indicates the annual rate of documented occurrences with all commercial fishing gear types in U.S. waters has been 5.2 (Waring *et al.* 2012). Based on the serious injury and mortality data for the past 10 years, we expect to see a range of zero to eight humpback whales seriously injured or killed each year as a result of U.S. fishing gear. Because serious injury or mortality could result from the seven fisheries, this Opinion assumes that serious injury or mortality could and would occur as a result of the seven fisheries.

Potential biological removal (PBR) for the Gulf of Maine humpback whale stock is 2.7 whales (Waring *et al.* 2012) which is higher than what was seen in the 2007-2011 stock assessment reports. As indicated above, while the annual average rate of documented serious injury/mortality events for humpback whales in sink gillnet gear is less than PBR (0.2 < 2.7), the overall annual rate of documented serious injury/mortality events with all U.S. commercial fishing gear for humpback whales is 4.4, which exceeds the PBR value of 2.7. 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 draft 2012 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. What we must consider in this Opinion is whether the continued operation of the seven fisheries over the next ten years will result in interactions with humpback whales that will result in serious injuries or mortalities that are likely to appreciably reduce the survival and recovery of the Gulf of Maine stock of humpback whales. If so, then we would have to determine if that appreciable reduction in survival and recovery for the Gulf of Maine stock resulted in an appreciable reduction in survival and recovery for humpback whales, which as previously noted, are listed as a single global species that is endangered throughout its range.

According to the latest final stock assessment report, the best abundance estimate for Gulf of Maine humpback whales was 823 animals and the minimum population estimate is 823 animals. The Gulf of Maine feeding population is estimated to be increasing at a rate of 6.5% for the period 1979-1991 (Barlow and Clapham, 1997). However, using data from 1992 through 2000, the population showed a lower growth rate of 0-4% (Clapham *et al.* 2003). A more precise estimate was not possible with available data; the lower estimate assumed a calf survival rate of 0.51 and the higher estimate was based on a calf survival rate of 0.875. The authors hypothesized that the apparent decline in growth rate during this later period could have resulted from a shift in humpback whale distribution to areas less sampled, a reduction in adult female survival, increased interbirth intervals or high mortality of first-year whales (such as off the Mid-Atlantic coast (Barco *et al.* 2002; Clapham *et al.* 2003). They considered reduced calf survival to be the most likely explanation and noted an apparent improvement after 1996. A subsequent study confirmed both low average reproductive rates and calf survival during much of that period (Robbins 2007). The average estimated calf survival rate for the period 2000-2005 (0.664, 95% CI: 0.517-0.784) fell between the values assumed by Clapham *et al.* (2003), and did not include neonatal mortality prior to arrival on the feeding ground (Robbins 2007). Regardless of the cause of lower calf survival between 1992 and 1995, Clapham *et al.* (2003) conclude that calf survival appears to have returned to near-previous levels beginning in 1996 and that it is likely that population growth is now comparable to that observed between 1979 and 1991 (6.5%). Given all of the available data, the draft 2012 stock assessment concludes that the Gulf of Maine humpback whale stock is steadily increasing in size. It is important to consider that the action being considered in this Opinion is not new, it is ongoing, and the Gulf of Maine humpback stock population has been increasing while the seven fisheries have continued to occur and continued to impact this stock. No changes to the fishery are being proposed that would increase the potential for interactions between the fishery and humpback whales.

The draft 2012 stock assessment concludes that the North Atlantic population of humpback whales overall had an estimated average population increase of 3.1% over the time period 1979-1993 (Waring *et al.* 2012; Stevick *et al.* 2003). Given that the GOM stock of humpback whales is increasing, it appears that the U.S. commercial fishery interactions are not currently threatening the survival of the Gulf of Maine stock of humpback whales, therefore it is logical to conclude that they are not threatening the survival of the overall stock of North Atlantic humpback whales.

The draft 2012 stock assessment concludes that human impacts (vessel collisions and entanglements) may be slowing recovery of humpback whale populations. In this Opinion, we must consider whether impacts associated with fishing authorized under the FMPs are likely to result in an appreciable reduction in the likelihood of recovery of humpback whales.

The goal of the 1991 Recovery Plan for the Humpback Whale (Plan) is to assist humpback whale populations to grow and to reoccupy areas where they were historically found. The long-term numerical goal of the Plan is to increase humpback whale populations to at least 60% of the existing number before commercial exploitation or of current environmental carrying capacity. With those levels undetermined, an intermediate goal was specified as a “doubling of extant populations within the next 20 years.”

The 1991 Plan used the 1986 population estimate for the Gulf of Maine feeding aggregation of humpback whales, which was 240 (95% CI = 147 to 333) (NMFS 1991b). As previously mentioned the best estimate of abundance for Gulf of Maine humpback whales is 823 animals (CV =0) and the current minimum population estimate is 823 animals (Waring *et al.* 2012).

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 10 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 yet have been observed in the population, and do not appear in the latest stock assessment report. The vertical line strategy developed under the ALWTRP, when implemented, will also benefit the population. While the details of the vertical line strategy are still being developed in consultation with the ALWTRT, there is a commitment to its implementation within a given time schedule.

As part of a large-scale assessment called More of North Atlantic Humpbacks (MoNAH) project, extensive sampling was conducted on humpbacks in the Gulf of Maine/Scotian Shelf region and the primary wintering ground on Silver Bank during 2004-2005. These data are being analyzed along with additional data from the U.S. Mid-Atlantic to estimate abundance and refine knowledge of population structure. This work is intended to update the Year of the North Atlantic Humpbacks (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 from management measurs, such as reductions in effort in the Northeast multispecies fisheries from Amendment 16 which was expected to resulte in a reduction of nearly 75% when compared to fishing effort and capacity in the early 1990’s (NEFMC 2009a). While some fishing effort may increase in the future as fisheries stocks respond to management measures to rebuild them, there are measures in place that will prevent overcapacity from redeveloping (*i.e*., nearly all U.S. Atlantic commercial fisheries are closed/limited access). Furthermore, as fish stocks increase, another possible outcome will be increased catches/landings with constant or even reduced fishing effort.

Specific downlisting criteria for humpback whales have not been developed. However, the estimated increases in the Gulf of Maine stock and the North Atlantic populations of humpback whales indicate that these populations are recovering despite continued interactions with commercial fisheries inside the U.S. EEZ. Additionally, there are indications of increasing abundance for the eastern and central North Pacific stocks (Waring *et al.* 2012) which are not impacted by the action under consideration in this Opinion.

The rate of humpback entanglements in fishing gear continues to be of concern to resource managers. The relatively new broad based gear modifications of the ALWTRP are expected to reduce the risk of SI/M due to humpback whale entanglement. The most recent data indicates the humpback whale population is steadily increasing despite the anthropogenic and cumulative effects previously discussed in this Opinion. While zero to eight interactions of humpback whales per year resulting in serious injury or mortality may occur under the continued authorization of the seven fisheries over the next ten years, the interaction level is not expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of this species.

### 9.3 Fin and Sei Whales

Serious injury and mortality entanglements of fin and sei whales have been documented but occur at a level below PBR for both species (Waring *et al.* 2012). This indicates that the level of serious injuries or mortalities of fin and sei whales attributable to U.S. commercial fisheries still allows these stocks to maintain population levels and growth rates needed to reach or maintain their optimum sustainable population. Additionally, effort in the seven fisheries is expected to be reduced and broad based gear modifications of the ALWTRP have been implemented. While interactions with fin and sei whales may occur under the continued authorization of the seven fisheries over the next ten years, the interaction level is not expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of these species.

### 9.4 NWA DPS of Loggerhead Sea Turtle

Based on information from Murray (2009a), Warden (2011a), and the STDN, we anticipate up to 483 loggerhead sea turtles from the NWA DPS will interact annually with gear utilized in the seven fisheries assessed in this Opinion. Loggerhead sea turtles that interact with gear used in these fisheries (which for the purposes of this Opinion includes gillnet, bottom trawl, hook gear, and trap/pot gear only) are those that are captured or entangled in the gear. An average of up to 269 loggerheads are expected to interact with gillnet gear annually based on the upper ends of the 95% CIs for the bycatch estimates by FMP group in Murray (2009a). In addition, an average of up to 213 loggerheads are expected to interact annually with bottom trawl gear, based on the upper ends of the 95% CIs for the bycatch estimates by FMP group in Warden (2011a). Also, up to one loggerhead is expected to interact annually with trap/pot gear in the black sea bass/scup fishery. Fifty-eight percent (156) of the annual interactions in gillnet gear and 47% (71) of the annual interactions in bottom trawl gear are expected to lead to serious injury or mortality, while the one loggerhead interaction in trap/pot gear could possibly be lethal. Therefore, up to 228 of the 483 loggerhead sea turtles that interact with these fisheries annually are expected to die or sustain serious injuries leading to death or failure to reproduce.

The lethal removal of up to 228 loggerhead sea turtles from the NWA DPS 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 actions (assuming all other variables remained the same). These lethal interactions would also result in a future reduction in reproduction as a result of lost reproductive potential, as some of these individuals would be females who would have survived other threats and reproduced in the future, thus eliminating each female individual's contribution to future generations. For example, an adult female loggerhead sea turtle can lay three or four clutches of eggs every two to four years, with 100 to 130 eggs per clutch. The annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. A reduction in the distribution of loggerhead sea turtles is not expected from lethal interactions attributed to the proposed actions. Because all the potential interactions are expected to occur at random throughout the action area and loggerheads generally have large ranges in which they disperse, the distribution of loggerhead sea turtles in the action area is expected to be unaffected.

Whether or not the reductions in NWA DPS loggerhead numbers and reproduction attributed to the proposed actions would appreciably reduce the likelihood of survival for loggerheads depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends (*i.e.*, whether the estimated reductions, when viewed within the context of the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* are to such an extent that adverse effects on population dynamics are appreciable). Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerheads require high survival rates throughout their life to maintain a population (Conant *et al.* 2009). In other words, late-maturing species are less tolerant of high rates of anthropogenic mortality. Conant *et al.* (2009) concluded that loggerhead natural growth rates are low, natural survival needs to be high, and even low(1-10%) to moderate (10-20%)mortality can drive the population into decline. Because recruitment to the adult population is slow, population modeling studies suggest even small increased mortality rates in adults and sub-adults could substantially impact population numbers and viability (Crouse *et al.* 1987; Crowder *et al.* 1994; Heppell *et al.* 1995; Chaloupka and Musick 1997).

Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of juveniles and adults in various fisheries and other marine activities. Conant *et al.* (2009) concluded that the results of their models (*i.e.*, predicted continued declines) are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the Northwest Atlantic. While significant progress has been made to reduce bycatch in some fisheries in certain parts of the loggerhead's range, and the results of new nesting trend analyses may indicate the positive effects of those efforts, notable fisheries bycatch persists. The question we are left with for this analysis is whether the effects of the proposed actions appreciably reduce survival and recovery, given the current status of the species and predicted population trajectories, as well as the many natural and human-caused impacts on sea turtles.

We may not see the long-term effects of the Deepwater Horizon oil release event and climate change on the population status and trends of loggerheads for several years.

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

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

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

The final revised recovery plan for loggerhead sea turtles in the Northwest Atlantic includes several 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, recovery criteria are framed in terms of both population parameters (Demographic Recovery Criteria) and the five listing factors (Listing Factor Recovery Criteria). For loggerheads, the nesting beach Demographic Recovery Criteria are specific to recovery units. The remaining criteria cannot be delineated by recovery unit because individuals in the recovery units mix in the marine environment; therefore, these criteria are applicable to all recovery units. Recovery criteria must be met for all recovery units in order for the species to be de-listed (NMFS and USFWS 2008). The Demographic Criteria for nests and nesting females were based on a time frame of one generation for U.S. loggerheads, defined in the recovery plan as 50 years. To be considered for delisting, each recovery unit will have recovered to a viable level and 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 the 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 may not reflect changes in the non-nesting population. This is because of the long time to maturity and the relatively small proportion of females that are reproducing on a nesting beach. 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 (NMFS and USFWS 2008).

As mentioned above, assuming some or all loggerhead sea turtles killed annually through interactions with these fisheries are females, the loss of female loggerhead sea turtles as a result of the proposed actions is expected to reduce the reproduction of loggerheads in the NWA DPS compared to the reproductive output of NWA DPS loggerheads in the absence of the proposed actions. In addition to being linked to survival, these losses are relevant to the Demographic Recovery Criteria for nests and nesting females. NMFS and USFWS (2008), Witherington *et al.* (2009), and TEWG (2009) provide comprehensive analyses of the status of the nesting assemblages within the NWA DPS using standardized data collected over 10-23 years. The results of these analyses, using different analytical approaches, were consistent—there had been a significant, overall nesting decline within this DPS. However, with the addition of nesting data from 2008 to 2010, which was not available at the time those analyses were conducted, the nesting trend from 1989 to 2010 is slightly negative, but the rate of decline is not statistically different from zero (76 FR 58868, September 22, 2011). Additionally, the range from the statistical analysis of the nesting trend includes both negative and positive growth (NMFS and USFWS 2008). The 2012 Florida index nesting number was the largest since 1998. The overall change in counts from 1989 to 2012 is positive.

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

Although limited information is available on the genetic makeup of loggerheads in an area as extensive as the action area, it is likely that loggerheads interacting with these seven fisheries originate from several, if not all of the recovery units. Cohorts from each of the five Northwest Atlantic nesting stocks have been documented to occur in the action area. Genetic analysis of samples collected from immature loggerheads captured in pound nets in the Pamlico-Albemarle Estuarine Complex in North Carolina between 1995-1997 indicated that 80% of the juveniles and subadults utilizing the foraging habitat originated from the south Florida nesting stock, 12% from the northern nesting stock, 6% from the Yucatán nesting stock, and 2% from other rookeries (including the Florida Panhandle, Dry Tortugas, Brazil, Greece, and Turkey nesting stocks) (Bass *et al.* 2004). In a separate study, genetic analysis of samples collected from loggerheads from Massachusetts to Florida also found that all five western Atlantic loggerhead stocks were represented (Bowen *et al.* 2004). However, earlier studies by Rankin-Baransky *et al.* (2001) and Witzell *et al.* (2002) indicated that only a few nesting stocks were represented along the U.S. Atlantic coast: south Florida (59% and 69% of the loggerheads sampled, respectively), northern (25% and 10%, respectively), and Mexico (16% and 20%, respectively). Most recently, Haas *et al.* (2008) used two approaches in identifying the contribution of each stock in U.S. Atlantic sea scallop fishery bycatch: an equal contribution from each stock or a weighted contribution by rookery sizes. When weighted by population size, Haas *et al.* (2008) found that 89% of the loggerheads captured in the U.S. Atlantic scallop fishery from 1996-2005 originated from the south Florida nesting stock, 4% were from the Mexican stock, 3% were from the northern (northeast Florida to North Carolina) stock, 1% were from the northwest Florida stock, and 0% were from the Dry Tortugas stock. The remaining 3% of loggerheads sampled were attributed to nesting stocks in Greece. Haas et al (2008) noted that these results should be interpreted with caution given the small sample size and resulting difficulties in precisely assigning rookery contributions to a particular mixed population. A re-analysis of loggerhead genetics data by the Atlantic Loggerhead TEWG has found that it is unlikely that U.S. fishing fleets are interacting with the Mediterranean DPS (LaCasella *et al.* In Review). Given that updated, more refined analyses are ongoing and the occurrence of Mediterranean DPS juveniles in U.S. Atlantic waters is rare and uncertain, if occurring at all, it is unlikely that individuals from the Mediterranean DPS would be present in the action area (Memorandum from Patricia A. Kurkul, Regional Administrator, to the Record, November 29, 2011). As a result, those records are excluded from our analysis and are reapportioned to the five Northwest Atlantic stocks which are expected to contribute to individuals in the action area. Note that when equal contributions of each stock were considered, Haas *et al.* (2008) found that the results varied from the weighted contributions but the south Florida nesting stock still contributed the majority of scallop fishery bycatch (63%).

The previously defined loggerhead nesting stocks do not share the exact delineations of the recovery units identified in the 2008 recovery plan. However, the PFRU encompasses the south Florida stock, the NRU is roughly equivalent to the northern nesting stock, the northwest Florida stock is included in the NGMRU, the Mexico stock is included in the GCRU, and the DTRU encompasses the Dry Tortugas stock. The available genetic analyses indicate the majority of bycatch in Northeast and Mid-Atlantic waters comes from the PFRU with smaller contributions from the other recovery units in the Northwest Atlantic DPS. However, the exact percentages of fisheries bycatch from specific nesting beaches and recovery units are not available at this time and may be variable from year to year. As a result, we are relying on the genetic analysis presented in Haas *et al.*

(2008), which is the most recent and one of the most comprehensive (in terms of the area from which samples were acquired) of the loggerhead genetics studies referenced above. The best available information indicates that the proportion of the interactions from each recovery unit is consistent with the relative sizes of the recovery units.The vast majority of the up to 228 loggerheads that are anticipated to be seriously injured or killed annually due to the seven fisheries assessed in this Opinion are likely to originate from the PFRU, with the remainder originating from the NRU, GCRU, NGMRU, and DTRU. Using the mean percent contributions in Haas *et al.* (2008) and then reapportioning the extra 3% of turtles that had been attributed to nesting stocks in Greece, we expect that 204 of the loggerheads killed or seriously injured will be from the PFRU, 8 from the NRU, 11 from the GCRU, 4 from the NGMRU, and one from the DTRU. Therefore, we conclude that none of the recovery units will be disproportionately impacted by interactions in these fisheries. Thus, genetic heterogeneity should be maintained in the species.

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

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. Estimates of the total loggerhead population in the Atlantic are not currently available. However, the AMAPPS aerial line transect surveys and sea turtle telemetry studies conducted along the U.S. Atlantic coast in the summer of 2010 provided preliminary regional abundance estimate of about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NEFSC 2011b). The estimate increases to approximately 801,000 (inter-quartile range of 521,000-1,111,000) when based on known loggerheads and a portion of unidentified sea turtle sightings. Also, a recent loggerhead population estimate prepared by Richards *et al.* (2011) using data from 2001-2010 states that the loggerhead adult female population in the Northwest Atlantic is 38,334 individuals (SD =2,287). They estimated adult female recovery unit sizes to range from a minimum of 258 females for the DTRU to a maximum of 45,048 females for the PFRU. 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 interacting with the fisheries are females and that all the interactions are of adults (a worst case scenario as far as reproductive value to the population), the loggerhead mortality as a result of these fisheries would result in the removal of 0.30% of the adult female loggerhead population in the Northwest Atlantic (114 out of 38,334). In general, while the loss of a certain number of individuals from a 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 the NWA DPS of loggerheads because the species is widely geographically distributed, it is not known to have low levels of genetic diversity, and there are tens to hundreds of thousands of individuals (and possibly more) in the DPS.

In determining whether the continued operation of the seven fisheries would reduce appreciably the likelihood of survival and recovery of loggerhead sea turtles, NMFS also 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 NWA DPS of loggerheads has not been 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.

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

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

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

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

Although the PVA is over four years old, uses data from 1989-2005, and models different effects of the fishery on loggerheads than what may occur presently, it is still appropriate as a standard for comparison in this Opinion as the current levels of loggerhead nesting in the Southeast United States (*i.e.*, the NRU and PFRU) are believed to be on a positive trend since 2008 when the trend was a long term decline for the time period assessed in the PVA. The PVA analysis done for the 2008 Opinion and our comparison of its results to the current status and trends of the NWA loggerhead DPS (in light of effects from these fisheries, other baseline activities, and climate change) supports the conclusion that continued operation of the seven fisheries will neither affect the number of nests and nesting females (Demographic Criteria #1) nor the trends in abundance on foraging grounds (Demographic Criteria #2) to the point where there is an appreciable reduction in the species' likelihood of recovery. Recovery is the process of removing threats so self-sustaining populations persist in the wild.

NMFS believes it is appropriate to use the results of the 2008 PVA to assess whether the seven fisheries as they currently operate will result in jeopardy for the NWA DPS of loggerhead sea turtles. Therefore, the loss of up to 228 individuals per year is unlikely to cause an appreciable reduction in the species' likelihood of survival and recovery. This is a NMFS determination based on the PVA results, it is not a determination of the PVA itself.

Even amidst ongoing threats to the species such as fishery mortality and climate change, the potential loss of 228 loggerheads annually from the Atlantic over the next ten years (and potentially beyond) is not likely to result in any appreciable decline to the NWA DPS. This is due to the large size of the current nesting population, the fact that the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts have been implemented and are underway to address threats.

### 9.5 Leatherback Sea Turtle

There have been several documented captures of leatherback sea turtles in gillnet, bottom trawl, and trap/pot gear utilized by the fisheries in the action area. Leatherback interactions with the fisheries are likely to continue given that the distribution of leatherbacks overlaps with areas where the gears are fished. From 2002 to 2011, there were four confirmed interactions of leatherback sea turtles with gillnet gear, two confirmed interactions with bottom otter trawl gear, and ten confirmed interactions with black sea bass trap/pot gear in the action area (NEFOP, ASM, and STDN databases). Based on these data, the bycatch of leatherback sea turtles in gillnet, trawl, or pot/trap gear within the action area is expected to occur, but likely at low levels.

Captures and/or entanglements of leatherback sea turtles in gillnet, bottom trawl, and trap/pot gear could result in death due to forced submergence, given that there are no regulatory controls on tow/soak times in these fisheries other than the 30-day maximum soak period for gillnets and trap/pot gear under the ALWTRP. Given that leatherbacks forage within the water column rather than on the bottom, interactions between leatherbacks and gillnet and trap/pot gear are expected to occur via the vertical lines and net panels. Interactions with bottom trawl gear are expected to occur when the gear is traveling through the water column versus on the bottom. As described in section 7.5, we anticipate up to eight leatherback sea turtle interactions annually with gillnet and bottom trawl gear used in the fishery, of which five are expected to be lethal. We also expect up to four annual interactions with black sea bass/scup trap/pot gear, which could be lethal or non-lethal.

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

The Leatherback TEWG estimated that there are between 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic (TEWG 2007). Of the five leatherback populations or groups of populations in the North Atlantic, three show an increasing or stable trend (Florida, Northern Caribbean, and Southern Caribbean). This includes the largest nesting population, located in the Southern Caribbean at Suriname and French Guiana. In 2001, the number of nests for Suriname and French Guiana was 60,000; this was one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). Of the remaining two populations, there was not enough information available on the West African population to conduct a trend analysis, while for the Western Caribbean, a slight decline in annual population growth rate was detected (TEWG 2007). An annual growth rate of 1.0 is considered a stable population; the growth rates of two nesting populations in the Western Caribbean were 0.98 and 0.96 (TEWG 2007). A stable trend in nesting suggests that leatherbacks are able to maintain current levels of nesting as well as current numbers of adult females despite on-going activities as described in the *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 previously. The result of which is that more female leatherbacks are maturing and subsequently nesting, and/or are surviving to an older age and producing more nests across their lifetime.

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

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

The recovery plan for Atlantic leatherback sea turtles (NMFS and USFWS 1992b) lists the following recovery objective, which is relevant to the proposed actions in this Opinion:

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

We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. In Puerto Rico, the main nesting areas are at Fajardo on the main island of Puerto Rico and on the island of Culebra. Between 1978 and 2005, nesting increased in Puerto Rico from a minimum of nine nests recorded in 1978 to 469-882 nests recorded each year between 2000 and 2005. Annual growth rate was estimated to be 1.1 with a growth rate interval between 1.04 and 1.12, using nest numbers between 1978 and 2005 (NMFS and USFWS 2007b). In the U.S. Virgin Islands, researchers estimated a population growth of approximately 13% per year at Sandy Point National Wildlife Refuge from 1994-2001. Between

1990 and 2005, the number of nests recorded has ranged from 143 (1990) to 1,008 (2001). The average annual growth rate was calculated as approximately 1.10 (with an estimated interval of 1.07 to 1.13) (NMFS and USFWS 2007b). In Florida, a Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 (1989) to 800-900 (early 2000s). Based on standardized nest counts made at Index Nesting Beach Survey sites surveyed with constant effort over time, there has been a substantial increase in leatherback nesting in Florida since 1989. The estimated annual growth rate was approximately 1.18 (with an estimated 95% CI of 1.1 to 1.21) (NMFS and USFWS 2007b).

Based on the information provided above, the loss of up to nine leatherback sea turtles annually in the Atlantic as a result of the continued operation of the fisheries will not appreciably reduce the likelihood of survival for 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 fisheries have no effects on leatherback sea turtles that occur outside of the Atlantic. Therefore, in light of other ongoing actions affecting leatherback sea turtles in the action area, the continued operation of the fisheries will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic. As a result, the proposed actions will not appreciably reduce the likelihood of survival of the species.

The annual loss of up to nine leatherback sea turtles, together with an increase in nesting, is not expected to affect the positive growth rate in the female population of leatherback sea turtles nesting in Puerto Rico, the U.S. Virgin Islands, and Florida. Therefore, the continued operation of the fisheries within the constraints of their FMPs will not appreciably reduce the likelihood of recovery for leatherback sea turtles in the Atlantic. Since the fisheries have no effects on leatherback sea turtles that occur outside of the Atlantic, their continued operation will not appreciably reduce the likelihood of recovery for the species.

### 9.6 Kemp's Ridley Sea Turtle

Kemp's ridley sea turtles have been documented to interact with both gillnet and bottom trawl gear in the action area. The distribution of Kemp's ridleys overlaps seasonally with the use of these gears and they are known to be captured in or entangled by gears used in several of the fisheries assessed in this Opinion, albeit at low levels. From 2001-2010 there were six confirmed captures of Kemp's ridley sea turtles in gillnet gear and two confirmed captures in bottom otter trawl gear in the action area (NEFOP database). Based on the observer data, we anticipate that up to three Kemp's ridley sea turtle interactions with gillnet gear and up to three interactions with bottom trawl gear will occur annually as a result of the continued operation of these fisheries. Of these fishery interactions, we anticipate that up to four will result in serious injury or mortality due to forced submergence or severe entanglement in the gear. Either male or female Kemp's ridleys may be captured/entangled in these fisheries since available information suggests that both sexes occur in the action area. All Kemp's ridleys interacting with these fisheries in the action area are expected to be immatures.

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

Since the mid-1980s, the number of nests observed at Rancho Nuevo and nearby beaches has increased 14%-16% per year (Heppell *et al.* 2005), allowing cautious optimism that the population is on its way to recovery. The total annual number of nests recorded at Rancho Nuevo and adjacent camps has exceeded 10,000 in recent years. Over 20,000 nests were recorded in 2009 at Rancho Nuevo and adjacent camps (J. Pena, GPZ, pers. comm.). From 2002 to 2009, a total of 771 Kemp's ridley nests were documented on the Texas coast. This is more than nine times greater than the 81 nests recorded over the previous 54 years from 1948-2001 (Shaver and Caillouet 1998; Shaver 2005), indicating an increasing nesting population in Texas. From 2005 to 2009, the number of nests from all monitored beaches indicate approximately 5,500 females are nesting each season in the Gulf of Mexico (NMFS *et al.* 2011). The observed increase in nesting of Kemp's ridley sea turtles suggests that the combined impact to Kemp's ridley sea turtles from on-going activities as described in the *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.

Heppell *et al.* (2005) predicted in a population model that the Kemp's ridley sea turtle population is expected to increase at least 12%-16% per year and that the population could attain at least 10,000 females nesting on Mexico beaches by 2015. NMFS (2011) contains an updated model which predicts that the population is expected to increase 19% per year and that the population could attain at least

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

It is likely that the Kemp's ridley sea turtle was the sea turtle species most affected by the Deepwater Horizon oil spill on a population level. In addition, the sea turtle strandings documented in 2010 and 2011 in Alabama, Louisiana, and Mississippi primarily involved Kemp's ridley sea turtles. Necropsy results indicated that mortality was caused by forced submergence, which is commonly associated with fishery interactions (77 FR 27413). Nevertheless, the effects on Kemp's ridley sea turtles from the proposed actions are not likely to appreciably reduce overall population numbers over time due to current population sizes, expected recruitment, and continuing strong nesting numbers (including, based on preliminary information, in 2011), even in light of the adverse impacts expected to have occurred from the Deepwater Horizon oil spill and the strandings documented in 2010 and 2011.

As described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to Kemp's ridley sea turtles. These include measures implemented to reduce the number and severity of Kemp's ridley sea turtle interactions in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries, the Mid-Atlantic scallop dredge fishery, and the Virginia pound net fishery. Since some of these regulatory measures have been in place for a number of years now, it is likely that current nesting trends reflect the benefit of these measures to Kemp's ridley sea turtles. Therefore, the current nesting trends for Kemp's ridleys are likely to continue to improve as a result of regulatory actions taken for these and other fisheries. There are no new known sources of serious injury or mortality for Kemp's ridley sea turtles other than potential impacts from the Deepwater Horizon oil spill.

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

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

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

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

The loss of up to four Kemp's ridleys annually is not expected to change the trend in increased nesting. Based on what we know about historical shrimp trawling effort (*i.e.*, that there has been much higher effort in the recent past), it is likely that large numbers of turtles were being impacted by shrimp trawls for the past decade or more. Despite this fact, the estimated population size of Kemp's ridleys has continued to increase. Therefore, in light of other ongoing actions affecting Kemp's ridley sea turtles in the action area, the continued operation of the fisheries within the constraints of their respective FMPs will not appreciably reduce the likelihood of recovery for the species.

### 9.7 Green Sea Turtle

Green sea turtles have been observed to interact with both gillnet and bottom trawl gear used in the seven fisheries that are the focus of this Opinion. From 2001 to 2010, there were 14 observed captures of green sea turtles in gillnet gear and one observed capture in bottom otter trawl gear in the New England and Mid-Atlantic regions (NEFOP database). Based on these observer data, we anticipate that up to four green sea turtle interactions with gillnet gear and up to three interactions with bottom trawl gear will occur annually as a result of the continued operation of these fisheries. Based on the lengths of soak/tow times for gillnet and bottom trawl fisheries in the action area, captures of green sea turtles in these gears could result in serious injuries or mortalities due to forced submergence. Currently there are no regulatory controls on tow times in these bottom trawl fisheries and the only restriction on gillnet soak times is the 30-day limit under the ALWTRP regulations. Serious injuries or mortalities could also occur as a result of entanglement in gillnet gear, which could hamper swimming, feeding, or surfacing behaviors and lead to asphyxiation or necrosis of body parts.

Shallow, coastal waters of the U.S. Atlantic from southern New England south are recognized as developmental habitat for green sea turtles after they enter the benthic environment (Musick and Limpus 1997; Morreale and Standora 2005; Makowski *et al.* 2006). In addition, nesting females have been documented to occur in action area waters as far north as Delaware, and nest in large numbers along the southeast coast of Florida. Thus, it is reasonable to expect that both benthic immature and sexually mature green sea turtles may be captured in gillnet and bottom trawl gear as a result of the continued operation of the fisheries.

The continued operation of these fisheries is anticipated to result in the annual serious injury or mortality of up to five green sea turtles—up to three in gillnet gear and up to two in bottom trawl gear. It is assumed that there is an equal chance of lethally capturing a male or female green sea turtle since available information suggests that both sexes occur in the action area. Lethal interactions would reduce the number of green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have otherwise survived to reproduce. For example, an adult female green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every two to four years with 110-115 eggs/nest, of which a small percentage is expected to survive to sexual maturity. A lethal capture of a female green sea turtle in gillnet or bottom trawl gear would remove reproductive output from the species. The anticipated lethal interactions are expected to occur anywhere in the action area, and green sea turtles generally have large ranges in which they disperse. Thus, no reduction in the distribution of green sea turtles is expected from these interactions. Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The five-year status review for green sea turtles states that of the seven green sea turtle nesting concentrations in the Atlantic Basin for which abundance trend information is available, all were determined to be either stable or increasing (NMFS and USFWS 2007d). That review also states that the annual nesting female population in the Atlantic basin ranges from 29,243-50,539 individuals. Additionally, the pattern of green sea turtle nesting shows biennial peaks in abundance, with a generally positive trend during the ten years of regular monitoring since the establishment of index beaches in Florida in 1989. An average of 5,039 green sea turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007d). Data from the index nesting beach program in Florida substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop was a temporary deviation from the normal biennial nesting cycle for green sea turtles, as 2010 saw an increase back to 8,426 nests on the index nesting beaches (FWC Index Nesting Beach Survey Database). Modeling by Chaloupka *et al.* (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population growing at 4.9% annually. 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 previously. The result of which is that more female green sea turtles are maturing and subsequently nesting, and/or are surviving to an older age and producing more nests across their lifetime.

We believe the proposed actions are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the green sea turtle. Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of green sea turtles would not be appreciably affected. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals. Since the abundance trend information for green sea turtles is clearly increasing while takes have been occurring, we believe the lethal interactions attributed to the proposed actions will not have any measurable effect on that trend. As described in the *Environmental*

*Baseline*, although the Deepwater Horizon oil spill is expected to have resulted in adverse impacts to green sea turtles, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from these fisheries would result in a detectable change in the population status of green sea turtles in the Atlantic. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions could be seen as reducing the likelihood of survival and recovery of the species.

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

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

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

Green sea turtle nest counts in Florida from 2001 to 2006 were documented as follows:

**Table 38** Green sea turtle nest counts in Florida (NMFS and USFWS 2007d)

| Year | Number of Nests |
|---|---|
| 2001 | 581 |
| 2002 | 9201 |
| 2003 | 2622 |
| 2004 | 3577 |
| 2005 | 9644 |
| 2006 | 4970 |
| Avg 2001-2006 | 5,039 |
| 2007 | 9455 |
| 2008 | 6385 |
| 2009 | 3000 |
| 2010 | 8426 |
| 2011 | 10701 |

Nest counts since 2006 have, on average, been even higher; thus, this recovery criterion continues to be met.

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

Based on the information provided above, the loss of up to five green sea turtles annually in the Atlantic as a result of the continued operation of the fisheries will not appreciably reduce the likelihood of survival for green sea turtles in the Atlantic given the increased nesting trend at the Atlantic nesting sites as well as measures that reduce the number of Atlantic green sea turtles that are injured and killed in the Atlantic (which should result in increases to the numbers of green sea turtles in the Atlantic that would otherwise have not occurred in the absence of those regulatory measures). The fisheries assessed in this Opinion have no effects on green sea turtles that occur outside of the Atlantic. Therefore, since the continued operation of the fisheries will not appreciably reduce the likelihood of survival of green sea turtles in the Atlantic, the proposed actions will not appreciably reduce the likelihood of survival for the species.

The annual loss of up to five green sea turtles, together with an increase in nesting, is not expected to measurably 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 fisheries will not appreciably reduce the likelihood of recovery for green sea turtles in the Atlantic. Since the fisheries have no effects on green sea turtles that occur outside of the Atlantic, and in light of other ongoing actions affecting green sea turtles in the action area, the continued operation of the seven fisheries within the constraints of their respective FMPs will not appreciably reduce the likelihood of recovery for the species.

### 9.8 Atlantic Sturgeon

Whether the reduction in numbers and reproduction from the loss of Atlantic sturgeon resulting from the proposed action would appreciably reduce the species likelihood of survival and recovery depends on how the changes in numbers and reproduction would affect the populations's growth rate, and whether the growth rate would allow the species to recover. For the population of each DPS to remain stable, a certain amount of spawning must occur within each DPS to offset deaths within each population. Two ways to measure spawning production are spawning stock biomass per recruit (SSB/R) and eggs per recruit (EPR). $EPR_{max}$ refers to the maximum number of eggs produced by a female Atlantic sturgeon over the course of its lifetime assuming no fishing mortality. Similarly, $SSB/R_{max}$ is the expected contribution a female Atlantic sturgeon would make to the total weight of the fish in a stock that are old enough to spawn during its lifetime over the course of its lifetime, assuming no fishing mortality. In both cases, as fishing mortality increases, the expected lifetime production of a female decreases from the theoretical maximum (i.e., $SSB/R_{max}$ or $EPR_{max}$) due to an increased probability the animal will be caught and therefore unable to achieve its maximum potential (Boreman 1997). Since the $EPR_{max}$ or $SSB/R_{max}$ for each individual within a population is the same, it is appropriate to talk about these parameters not only for individuals but for populations as well.
Goodyear (1993) suggests that maintaining a SSB/R of at least 20% of $SSB/R_{max}$ would
allow a population to remain stable (i.e., retain the capacity for survival). Boreman *et al.* (1984) indicated that maintaining a SSB/R of at least 50% of $SSB/R_{max}$ would be an appropriate target for rebuilding (i.e., recovery). Boreman (1997) indicates that since stock biomass and egg production are typically linearly correlated it is appropriate to apply the 20% (Goodyear 1993) and 50% (Boreman 1997) thresholds directly to EPR estimates.

Boreman (1997) reported adult female Atlantic sturgeon in the Hudson River could have likely sustained a fishing mortality rate of 14% and still retained enough spawners for the population to remain stable (i.e., maintain an EPR of at least 20% of $EPR_{max}$). Additionally, a fishing mortality rate of 5% corresponds to maintaining an EPR of at least 50% of $EPR_{max}$ (Boreman 1997). These fishing mortality rates are specific to adult female spawners. Since estimates of fishing mortality rates that would equal 20% and 50% of $EPR_{max}$ are not available for any of the five Atlantic sturgeon DPS, the information on the Hudson River is the best available. While we have some limited information on male to female ratios for the Hudson River (Erickson *et al*. 2011; Kahnle *et al*. 2007; Pekovitch 1979), we do not know the current sex ratio for adult or subadult sturgeon for any of the five Atlantic sturgeon DPSs. In the absence of this information, we chose to evaluate our anticipated takes of all adults against these female-specific fishing mortality rates because we believe doing so is conservative toward the species.

As noted previously in the Environmental Baseline of this Opinion, we believe equal portions of Atlantic sturgeon bycatch in fisheries may occur in federal fisheries and state fisheries. For purposes of this Opinion, we are including takes from state fisheries andfederal fisheries in the fishing mortality rates (F) throughout the assessment. We have grouped fishing mortality of Atlantic sturgeon equally within two categories: federal fisheries and state fisheries. Based on this assumption, we would anticipate each Atlantic sturgeon DPS could sustain a federal fishing mortality of approximately 7% (i.e., 50% of the Boreman 1997 14% threshold) and still retain enough spawners for the population to remain stable (i.e., maintain at least 20% of $EPR_{max}$). Likewise, we would anticipate each Atlantic sturgeon DPS could sustain a federal fishing mortality of approximately 2.5% (i.e., 50% of the Boreman 1997 5% threshold) and still retain enough spawners for the population to rebuild (i.e., maintain at least 50% of $EPR_{max}$).

We have considered the best available information to determine from which DPSs individuals that will be killed are likely to have originated. Using the genetic mixed stock analysis explained above, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: Gulf of Maine 11%; NYB 51%; Chesapeake Bay 13%; Carolina 2%; and South Atlantic 22%. Given these percentages, we expect that up to 36 of the Atlantic sturgeon mortalities will be fish that originate from Gulf of Maine DPS; up to 162 from the New York Bight DPS, up to 42 from the Chesapeake Bay DPS, up to 7 from the Carolina DPS; and up to 70 from the South Atlantic DPS.

Because the federal fisheries that are known to interact with Atlantic sturgeon include some fisheries managed through FMPs authorized by the NMFS Southeast Regional Office (SERO), we have evaluated the authorized incidental take and estimated incidental take of Atlantic sturgeon in the biological opinions finalized since the ESA listing of Atlantic sturgeon DPS. The authorized incidental take levels are listed in Table 19 (Section 5.1.2).

Because the MSA genetic percentages for Atlantic sturgeon DPSs have changed since the time the SERO biological opinions were formed for the Southeastern U.S. Shrimp Fisheries and the Atlantic shark fisheries managed under the Consolidated HMS FMP, we have converted the estimated take numbers from those Opinions to reflect the latest MSA genetic percentages data provided in 2012 (Table 39-Table 42).

**Table 39** Estimated number of Atlantic sturgeon interactions in the Southeastern U.S. Shrimp Fishery recalculated using updated 2012 MSA genetic percentages.

| | Estimated Interactions in Otter Trawl Gear | Estimated Atlantic sturgeon Escaping through TEDs in Otter Trawl Gear | Estimated Captures of Atlantic Sturgeon in Otter Trawl Gear* | Estimated Mortalities of Atlantic Sturgeon Interacting with Otter Trawl Gear* | Estimated Captures of Atlantic Sturgeon in Try Net Gear | Estimated Mortalities of Atlantic Sturgeon Interacting with Try Net Gear | Total Estimated Captures in Otter Trawl and Try Net Gear | Total Estimated Mortalities in Otter Trawl and Try Net Gear |
|---|---|---|---|---|---|---|---|---|
| **Total** | 570 | | | | 21** | 0 | | |
| **GOM (11%)** | 63 | 55 | 8 | 1 | 2 | 0 | 10 | 1 |
| **NYB (51%)** | 291 | 253 | 38 | 4 | 11 | 0 | 49 | 4 |
| **CB 13%)** | 74 | 64 | 10 | 1 | 3 | 0 | 13 | 1 |
| **Carolina (2%)** | 11 | 10 | 1 | 1* | 1 | 0 | 2 | 1 |
| **SA (22%)** | 125 | 109 | 16 | 2 | 5 | 0 | 21 | 2 |
| **Canada (1%)** | 6 | 5 | 1 | 1* | 1 | 0 | 2 | 1 |

* Estimated mortalities were rounded up to the next whole number to be conservative to the species.
**The sum of the DPS estimated capture values do not equal the estimated total because the Canadian fish and Carolina DPS percentages were less than one and were rounded up to one to be conservative to the species.

**Table 40** Estimated number of Atlantic sturgeon captures in the Atlantic shark fisheries recalculated using updated 2012 MSA genetic percentages

| | **Estimated Captures in Sink Gillnet Gear** | **Estimated Mortalities of Atlantic Sturgeon Interacting with Sink Gillnet Gear*** |
|---|---|---|
| **Total** | 108 | |
| **GOM (11%)** | 12 | 3 |
| **NYB (51%)** | 55 | 11 |
| **CB 13%)** | 14 | 3 |
| **Carolina (2%)** | 2 | 1 |
| **SA (22%)** | 24 | 5 |
| **Canada (1%)** | 1 | 1 |

*Estimated mortalities were rounded up to the next whole number to be conservative to the species.This means the estimated total lethal interaction values do not equal 20% of the estimated total captures in sink gillnet gear.

**Table 41** Anticipated Atlantic sturgeon adult and adult equivalent mortalities in the Southeast U.S. Shrimp Fisheries

| | | Dead Encounters by Life Stage | | | |
|---|---|---|---|---|---|
| | **Total Estimated Mortalities in Otter Trawl and Try Net Gear** | **25% adult** | **75% subadult** | **Dead Encounters: Subadults converted to Adult Equivalents** | **Dead Encounters: Adults Plus Adult Equivalents*** |
| **GOM (11%)** | 1 | 0.25 | 0.75 | 0.36 | 1 |
| **NYB (51%)** | 4 | 1 | 3 | 1.44 | 3 |
| **CB (13%)** | 1 | 0.25 | 0.75 | 0.36 | 1 |
| **Carolina (2%)** | 1 | 0.25 | 0.75 | 0.36 | 1 |
| **SA (22%)** | 2 | 0.5 | 1.5 | 0.72 | 2 |
| **Canada (1%)** | 1 | 0.25 | 0.75 | 0.36 | 1 |

*Estimated mortalities were rounded up to the next whole number to be conservative to the species.

**Table 42** Anticipated Atlantic sturgeon adults and adult equivalents mortalities in the Southeast shark fisheries managed under the Consolidated HMS FMP

| | | Dead Encounters by Life Stage | | | |
|---|---|---|---|---|---|
| | **Total Estimated Mortalities in Sink Gillnet Gear** | **25% adult** | **75% subadult** | **Dead Encounters: Subadults converted to Adult Equivalents** | **Dead Encounters: Adults Plus Adult Equivalents*** |
| **GOM (11%)** | 3 | 0.75 | 2.25 | 1.08 | 2 |
| **NYB (51%)** | 11 | 2.75 | 8.25 | 3.96 | 7 |
| **CB (13%)** | 3 | 0.75 | 2.25 | 1.08 | 2 |
| **Carolina (2%)** | 1 | 0.25 | 0.75 | 0.36 | 1 |
| **SA (22%)** | 5 | 1.25 | 3.75 | 1.8 | 4 |
| **Canada** | 1 | 0.25 | 0.75 | 0.36 | 1 |

| | | | | | |
|---|---|---|---|---|---|
| **(1%)** | | | | | |

*Estimated mortalities were rounded up to the next whole number to be conservative to the species.

Because our analysis for each DPS requires comparing impacts to Atlantic sturgeon adults and adult equivalents, we have reproduced the table for population estimates per life stage and added a column for adult equivalents.

**Table 43** Summary of calculated population estimates, including adult equivalents, based upon the NEAMAP Survey swept area assuming 50% efficiency

| DPS | Estimated Ocean Population Abundance | Estimated Ocean Population of Adults | Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries) | Estimated Ocean Population of Adult Equivalents | Estimated Ocean Population of Adults/Adult Equivalents |
|---|---|---|---|---|---|
| **GOM (11%)** | 7,455 | 1,864 | 5,591 | 2,684 | 4,548 |
| **NYB (51%)** | 34,566 | 8,642 | 25,925 | 12,444 | 21,086 |
| **CB (13%)** | 8,811 | 2,203 | 6,608 | 3,172 | 5,375 |
| **Carolina (2%)** | 1,356 | 339 | 1,017 | 488 | 827 |
| **SA (22%)** | 14,911 | 3,728 | 11,183 | 5,368 | 9,096 |
| **Canada (1%)** | 678 | 170 | 509 | 244 | 414 |

### 9.8.1 GOM DPS

The proposed action may result in up to 285 Atlantic sturgeon takes from the GOM DPS annually. As shown in Section 7.6.2, we estimated those takes will likely result in mortalities of 9.06 adult and 25.57 subadult Atlantic sturgeon. Converting the 25.57 subadults to adult equivalents (25.57 x 0.48) produced 12.27 fish. Adding the 12.27 adult equivalents to the 9.06 adults produced an annual average of 21.33

Atlantic sturgeon adult/adult equivalent mortalities in the GOM DPS. Since a portion of a fish cannot be taken, we rounded the 21.33 to 22.

We anticipate that 22 adults/adult equivalents from the GOM DPS may be lethally taken by the proposed action. The Opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of up to one adult/adult equivalent from the GOM DPS annually. The Opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of one lethal take of an adult/adult equivalent from the GOM DPS annually. The Opinion for the Atlantic shark fisheries managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of up to two adults/adult equivalents from the GOM DPS annually. Together, we anticipate that up to 26 adults/adult equivalents Atlantic sturgeon from the GOM DPS may be removed annually because of federal fisheries, or 0.57% of the adult/adult equivalent population in the GOM DPS (i.e. 4,548). This 0.57% is below the estimated 7% federal fishing mortality rate we believe the population could likely withstand and still maintain 20% of $EPR_{max}$.

The proposed action may result in the annual average removal of 22 Atlantic sturgeon that would have been reproductive adults from the GOM DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the GOM DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where GOM DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by GOM DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by GOM DPS subadults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 22 adult/adult equivalent GOM DPS Atlantic sturgeon over the ten-year period considered in this Opinion, will not appreciably reduce the likelihood of survival of the GOM DPS (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect GOM DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

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

As noted previously, Boreman (1997) suggested maintaining an EPR of at least 50% of $EPR_{max}$ would be appropriate to rebuild a species with life history characteristics like Atlantic sturgeon. Boreman (1997) estimated an EPR of at least 50% of $EPR_{max}$ could be maintained for the Hudson River population if fishing mortality remained at or below 5%. If we follow the same assumptions noted previously regarding a 50:50 split between Atlantic sturgeon bycatch in state fisheries andfederal fisheries, the Atlantic sturgeon bycatch from the proposed action and other federal fisheries would have to remain below 2.5% (i.e., 50% of the Boreman 1997 5% threshold) to maintain enough spawners for the population to rebuild. Previously we estimated that the proposed action, in conjunction with other federal fisheries, likely removes 0.57% of adults/adult equivalents in the GOM DPS. This estimate is below the 2.5% threshold we believe is necessary to maintain an EPR of at least 50% of $EPR_{max}$.

As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

Within the ten-year period considered for this consultation, the proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of GOM DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of GOM DPS Atlantic sturgeon. The proposed action will not utilize GOM DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than 22 adult/adult equivalent GOM DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the GOM DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of GOM DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the GOM DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the mortality of 22 adult/adult equivalent GOM DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

### 9.8.2 NYB DPS

The proposed action may result in up to 1,317 Atlantic sturgeon takes from the NYB DPS annually. As shown above, we estimated those takes will likely result in mortalities of 41.99 adult and 118.57 subadult Atlantic sturgeon. Converting the 118.57 subadults to adult equivalents (118.57 x 0.48) produces 56.91 fish. Adding the 56.91 adult equivalents to the 41.99 adults produces an annual average of 98.90 Atlantic sturgeon adult/adult equivalent mortalities in the NYB DPS. Since a portion of a fish cannot be taken we have rounded the 98.90 to 99.

We anticipate that 99 adults/adult equivalents from the NYB DPS may be lethally taken by the proposed action. The Opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of up to one adult/adult equivalent from the NYB DPS annually. The Opinion for the Southeastern U.S. shrimp trawl fishery provides for an average lethal takes of up to three adults/adult equivalents from the NYB DPS annually. The Opinion for the Atlantic shark fisheries managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of up to seven adults/adult equivalents from the NYB DPS annually. Together, we anticipate a total of up to 110 adults/adult equivalents Atlantic sturgeon from the NYB DPS may be removed annually because of federal fisheries, or 0.52% of the adult/adult equivalent population in the NYB DPS (i.e., 21,086). This 0.52% is below the estimated 7% federal fishing mortality rate we believe the population could likely withstand and still maintain 20% of $EPR_{max}$.

The proposed action may result in the anticipated annual average removal of 99 Atlantic sturgeon that would have been reproductive adults from the NYB DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the NYB DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where NYB DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by NYB DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by NYB DPS subadults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 99 adult/adult equivalent NYB DPS Atlantic sturgeon over the ten-year period considered in this Opinion will not appreciably reduce the likelihood of survival of the NYB DPS (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect NYB DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that NYB DPS Atlantic sturgeon will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery.

As noted previously, Boreman (1997) suggested maintaining an EPR of at least 50% of $EPR_{max}$ would be appropriate to rebuild a species with life history characteristics like Atlantic sturgeon. Boreman (1997) estimated an EPR of at least 50% of $EPR_{max}$ could be maintained for the Hudson River population if fishing mortality remained at or below 5%. If we follow the same assumptions noted previously regarding a 50:50 split between Atlantic sturgeon bycatch in state fisheries and federal fisheries, the Atlantic sturgeon bycatch from the proposed action and other federal fisheries would have to remain below 2.5% (i.e., 50% of the Boreman 1997 5% threshold) to maintain enough spawners for the population to rebuild. Previously we estimated that the proposed action, in conjunction with other federal fisheries, likely removes 0.52% of adults/adult equivalents in the NYB DPS. This estimate is below the 2.5% threshold we believe is necessary to maintain an EPR of at least 50% of EPRmax.

As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

Within the ten-year period considered for this consultation, the proposed action is not expected to modify, curtail or destroy the range of the species since it will result in a small reduction in the number of NYB DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of NYB DPS Atlantic sturgeon. The proposed action will not utilize NYB DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than 99 adult/adult equivalent NYB DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the NYB DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of NYB DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the NYB DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the mortality of 99 adult/adult equivalent NYB DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

### 9.8.3 CB DPS

The proposed action may result in up to 337 Atlantic sturgeon takes from the CB DPS annually. As shown in Section 7.6.2, we estimated those takes will likely result in lethal takes of 10.73 adult and 30.23 subadult Atlantic sturgeon. Converting the 30.23 subadults to adult equivalents (30.23 x 0.48) produces 14.51 fish. Adding the 14.51 adult equivalents to the 10.73 adults produces an annual average of 25.24 Atlantic sturgeon adult/adult equivalent mortalities in the CB DPS. Since a portion of a fish cannot be taken we have rounded 25.24 to 26.

We anticipate that 26 adults/adult equivalents from the CB DPS may be lethally taken by the proposed action. The Opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of up to one adult/adult equivalent from the CB DPS annually. The Opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of one lethal take of an adult/adult equivalent from the CB DPS annually. The Opinion for Atlantic shark fisheries managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of up to two adults/adult equivalents from the CB DPS annually. Together, we anticipate that a total of up to 30 adults/adult equivalents Atlantic sturgeon from the CB DPS may be removed annually because of federal fisheries, or 0.56% of the adult/adult equivalent population in the CB DPS (i.e. 5,375). This 0.56% is below the estimated 7% federal fishing mortality rate we believe the population could likely withstand and still maintain 20% of EPRmax.

The proposed action may result in the anticipated annual average removal of 26 Atlantic sturgeon that would have been reproductive adults from the CB DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the CB DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where CB DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by CB DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by CB DPS subadults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 26 adult/adult equivalent CB DPS Atlantic sturgeon over the ten-year period considered in this Opinion, will not appreciably reduce the likelihood of survival of the CB DPS (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect CB DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that CB DPS Atlantic sturgeon will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery.

As noted previously, Boreman (1997) suggested maintaining an EPR of at least 50% of $EPR_{max}$ would be appropriate to rebuild a species with life history characteristics like Atlantic sturgeon. Boreman (1997) estimated an EPR of at least 50% of $EPR_{max}$ could be maintained for the Hudson River population if fishing mortality remained at or below 5%. If we follow the same assumptions noted previously regarding a 50:50 split between Atlantic sturgeon bycatch in state fisheries and federal fisheries, the Atlantic sturgeon bycatch from the proposed action and other federal fisheries would have to remain below 2.5% (i.e., 40% of the Boreman 1997 5% threshold) to maintain enough spawners for the population to rebuild. Previously we estimated the proposed action, in conjunction with other federal fisheries, likely removes 0.56% of adults/adult equivalents in the CB DPS. This estimate is below the 2.5% threshold we believe is necessary to maintain an EPR of at least 50% of EPRmax.

As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

Within the ten-year period considered for this consultation, the proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of CB DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of CB DPS Atlantic sturgeon. The proposed action will not utilize CB DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than 26 adult/adult equivalent CB DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the CB DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of CB DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the CB DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the mortality of 26 adult/adult equivalent CB DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

### 9.8.4 Carolina DPS

The proposed action may result in up to 52 Atlantic sturgeon takes from the Carolina DPS annually. As shown in Section 7.6.2, we estimated those takes will likely result in lethal takes of 1.65 adult and 24.37 subadult Atlantic sturgeon. Converting the 24.37 subadults to adult equivalents (24.37 x 0.48) produced 11.70 fish. Adding the 11.70 adult equivalents to the 1.65 adults produced an annual average of 13.35 Atlantic sturgeon adult/adult equivalent mortalities in the Carolina DPS. Since a portion of a fish cannot be taken we have rounded 13.35 to 14.

We anticipate that 14 adults/adult equivalents from the Carolina DPS may be lethally taken by the proposed action. The Opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of up to one adult/adult equivalent from the Carolina DPS annually. The Opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of one lethal take of an adult/adult equivalent from the Carolina DPS annually. The Opinion for Atlantic shark fisheries managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of up to one adult/adult equivalent from the Carolina DPS annually. Together, we anticipate that a total of up to 17 adults/adult equivalents Atlantic sturgeon from the Carolina DPS may be removed annually because of federal fisheries, or 2.1% of the adult/adult equivalent population in the Carolina DPS (i.e., 827). This 2.1% is below the estimated 7% federal fishing mortality rate we believe the population could likely withstand and still maintain 20% of $EPR_{max}$.

The proposed action may result in the anticipated annual average removal of 14 Atlantic sturgeon that would have been reproductive adults from the Carolina DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the Carolina DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where Carolina DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by Carolina DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by Carolina DPS subadults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 14 adult/adult equivalent Carolina DPS Atlantic sturgeon over the ten-year period considered in this Opinion, will not appreciably reduce the likelihood of survival of the Carolina DPS (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect Carolina DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that Carolina DPS Atlantic sturgeon will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery.

As noted previously, Boreman (1997) suggested maintaining an EPR of at least 50% of $EPR_{max}$ would be appropriate to rebuild a species with life history characteristics like Atlantic sturgeon. Boreman (1997) estimated an EPR of at least 50% of $EPR_{max}$ could be maintained for the Hudson River population if fishing mortality remained at or below 5%. If we follow the same assumptions noted previously regarding a 50:50 split between Atlantic sturgeon bycatch in state fisheries and federal fisheries the Atlantic sturgeon bycatch from the proposed action and other federal fisheries would have to remain below 2.5% (i.e., 50% of the Boreman 1997 5% threshold) to maintain enough spawners for the population to rebuild. Previously we estimated that the proposed action, in conjunction with other federal fisheries, likely removes 2.1% of adults/adult equivalents in the Carolina DPS. This estimate is below the 2.5% threshold we believe is necessary to maintain an EPR of at least 50% of $EPR_{max}$.

As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

Within the ten-year period considered for this consultation, the proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of Carolina DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of Carolina DPS Atlantic sturgeon. The proposed action will not utilize Carolina DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than 14 adult/adult equivalent Carolina DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the Carolina DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of Carolina DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the Carolina DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the mortality of 14 adult/adult equivalent Carolina DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

### 9.8.5 SA DPS

The proposed action may result in up to 569 Atlantic sturgeon takes from the SA DPS annually. As shown in Section 7.6.2, we estimated those takes will likely result in lethal takes of 17.95 adult and 51.15 subadult Atlantic sturgeon. Converting the 51.15 subadults to adult equivalents (51.15 x 0.48) produced 24.55 fish. Adding the 24.55 adult equivalents to the 17.95 adults produced an annual average of 42.50 Atlantic sturgeon adult/adult equivalent mortalities in the SA DPS. Since a portion of a fish cannot be taken we have rounded 42.50 to 43.

We anticipate that 43 adults/adult equivalents from the SA DPS may be lethally taken by the proposed action. The Opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of up to one adults/adult equivalents from the SA DPS annually. The Opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of two lethal takes of adults/adult equivalents from the SA DPS annually. The Opinion for Atlantic shark fisheries managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of up to four adults/adult equivalents from the SA DPS annually. Together, we anticipate that a total of up to 50 adults/adult equivalents Atlantic sturgeon from the

SA DPS may be removed annually because of federal fisheries, or 0.55% of the adult/adult equivalent population in the SA DPS (i.e., 9,096). This 0.55% is below the estimated 7% federal fishing mortality rate we believe the population could likely withstand and still maintain 20% of EPRmax. We have chosen to compare the number of fish that may be removed annually because of federal fishing to the SA DPS adult population estimate and not the adults plus adult equivalent population estimate for the SA DPS to be conservative for the species.

The proposed action may result in the anticipated annual average removal of 43 Atlantic sturgeon that would have been reproductive adults from the SA DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the SA DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where SA DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by SA DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by SA DPS subadults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 43 adult/adult equivalent SA DPS Atlantic sturgeon over the ten-year period considered in this Opinion will not appreciably reduce the likelihood of survival of the SA DPS (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect SA DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that SA DPS Atlantic sturgeon will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery.

As noted previously, Boreman (1997) suggested maintaining an EPR of at least 50% of $EPR_{max}$ would be appropriate to rebuild a species with life history characteristics like Atlantic sturgeon. Boreman (1997) estimated an EPR of at least 50% of $EPR_{max}$ could be maintained for the Hudson River population if fishing mortality remained at or below 5%. If we follow the same assumptions noted previously regarding a 50:50 split between Atlantic sturgeon bycatch in state fisheries and federal fisheries the Atlantic sturgeon bycatch from the proposed action and other federal fisheries would have to remain below 2.5% (i.e. which is 50% of the Boreman 1997 5% threshold) to maintain enough spawners for the population to rebuild. Previously we estimated the proposed action, in conjunction with other federal fisheries, likely removes 0.55% of adults/adult equivalents in the SA DPS. This estimate is below the 2.5% threshold we believe is necessary to maintain an EPR of at least 50% of $EPR_{max}$.

As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

Within the ten-year period considered for this consultation, the proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of SA DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of SA DPS Atlantic sturgeon. The proposed action will not utilize SA DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than 43 adult/adult equivalent SA DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the SA DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of SA DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the SA DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the mortality of 43 adult/adult equivalent SA DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

### 9.9 GOM DPS Atlantic Salmon

Atlantic salmon have been observed to interact with both gillnet and bottom trawl gear used in the seven fisheries that are the focus of this Opinion. From 1989 through September 2012, there were nine observed captures of Atlantic salmon in gillnet gear and four observed captures in bottom otter trawl gear in the New England and Mid-Atlantic regions (NEFOP database). Based on these observer data, we anticipate up to one interaction with gillnet gear and one interaction with bottom trawl gear will occur on average annually as a result of the continued operation of these fisheries.

GOM DPS smolts generally enter the sea in May, and follow direct routes out of the coastal environment into the ocean (Hyvarinen *et al*. 2006; Lacroix and McCurdy 1996; Lacroix *et al*. 2004, 2005). Studies suggest that post-smolts move near the coast in migration corridors closely related to surface currents (Hyvarinen *et al*. 2006; Lacroix and McCurdy 1996; Lacroix *et al*. 2004). North American post-smolts appear to have a near-shore distribution (Friedland *et al*. 2003), and move to the Labrador Sea and off of the west coast of Greenland in the late summer to autumn of their first year (Reddin 1985; Reddin and Short 1991; Reddin and Friedland 1993). The salmon located off Greenland are composed of both 1SW fish and MSW fish, and includes immature salmon from both North American and European stocks (Reddin 1988; Reddin *et al*. 1988). In the spring, North American post-smolts are generally located in the Gulf of St. Lawrence, off the coast of Newfoundland, and on the east coast of the Grand Banks (Reddin 1985; Dutil and Coutu 1988; Ritter 1989; Reddin and Friedland 1993; and Friedland *et al*. 1999). Some salmon may remain at sea for another year or more before maturing, overwintering in the area of the Grand Banks before returning to their natal rivers to spawn (Reddin and Shearer 1987). Part of their migratory pattern overlaps with the action area at times when the seven fisheries are active.

Lethal takes are expected to occur on average every three years in gillnet gear and on average every two years in bottom trawl gear as a result of the continued operation of these fisheries. Lethal interactions would reduce the number of GOM DPS Atlantic salmon, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have otherwise survived to reproduce. For example, an adult 2SW female Atlantic salmon can produce a total of 1,500 to 1,800 eggs per kilogram of body weight, yielding an average of 7,500 eggs (Baum and Meister 1971), of which a small percentage is expected to survive to sexual maturity. A lethal capture of an adult female GOM DPS Atlantic salmon in gillnet or bottom trawl gear would likely remove this level of reproductive output from the species. The anticipated lethal interactions are expected to occur anywhere in the action area, though are most likely to occur in the Gulf of Maine and Georges Bank areas. Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The most recent data available on the population trend of Atlantic salmon indicate that their abundance within the range of the GOM DPS has been generally declining since the 1800s (Fay *et al.* 2006). Contemporary estimates of abundance for the entire GOM DPS have rarely exceeded 5,000 individuals in any given year since 1967 (Fay *et al.* 2006), and appear to have stabilized at very low levels since 2000. After a period of slow population growth between the 1970s and the early 1980s, adult returns of salmon in the GOM DPS peaked around 1985 and declined through the 1990s and early 2000s. Adult returns have been increasing again over the last few years. The population growth observed in the 1970s is likely attributable to favorable marine survival and increases in hatchery capacity, particularly from GLNFH that was constructed in 1974. Marine survival remained relatively high throughout the 1980s, and salmon populations in the GOM DPS remained relatively stable until the early 1990s. In the early 1990s, marine survival rates decreased, leading to the declining trend in adult abundance observed throughout 1990s and early 2000s. The increase in the abundance of returning adult salmon observed between 2008 and 2011 may be an indication of improving marine survival

Adult returns for the GOM DPS remain well below conservation spawning escapement (CSE) goals that are widely used (ICES 2005) to describe the status of individual Atlantic salmon populations. When CSE goals are met, Atlantic salmon populations are generally self-sustaining. When CSE goals are not met (i.e., less than 100%), populations are not reaching full potential; and this can be indicative of a population decline. For all GOM DPS rivers in Maine, current Atlantic salmon populations (including hatchery contributions) are well below CSE levels required to sustain themselves (Fay *et al.* 2006), which is further indication of their poor population status.

The observed declines in Atlantic salmon suggests that the combined impacts from ongoing activities described in the *Environmental Baseline*, *Cumulative Effects*, and the *Status of Listed Species* (including those activities that occur outside of the action area of this Opinion) are continuing to cause the population to deterioriate.

We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the GOM DPS Atlantic salmon. Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of Atlantic salmon would not be affected in any detectable way. For the population to remain stable, Atlantic salmon must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals from successful reproduction of Atlantic salmon that were not seriously injured or killed in the fisheries. While the abundance trend information for Atlantic salmon is either stable or declining, we believe the very small numbers of lethal interactions attributed to the proposed actions (up to one per year) will not have any measurable effect on that trend.

As also described in the *Environmental Baseline*, a number of actions are being taken to help Atlantic salmon recover. These include hatchery supplementation; removing dams or providing fish passage; improving road crossings that block passage or degrade stream habitat; protecting riparian corridors along rivers; reducing the impact of irrigation water withdrawals; limiting effects of recreational and commercial fishing; reducing the effects of finfish aquaculture; outreach and education activities; and research focused on better understanding the threats to Atlantic salmon and developing effective restoration strategies.

The draft 2010 recovery framework for Atlantic salmon has as its objectives to increase abundance, distribution, ecosystem function, and genetic diversity of the species. To support these objectives, a five-prong strategy was developed:

Strategy A: Increase Marine and Estuarine Survival
Strategy B: Increase Connectivity
Strategy C: Maintain Genetic Diversity through the Conservation Hatchery
Strategy D: Increase Adult Spawners through the Conservation Hatchery
Strategy E: Increase Adult Spawners through the Freshwater Production of Smolts

Improving the survival of Atlantic salmon in the marine environment is an important part of meeting the objective of GOM DPS Atlantic salmon recovery. However, there is no indication that activities of the seven fisheries are considered a threat to Atlantic salmon recovery. Therefore, we believe that the loss of up to one GOM DPS Atlantic salmon per year as a result of the continued operation of the seven fisheries will not reduce the likelihood of survival and recovery the GOM DPS of Atlantic salmon.

## 10.0 Conclusion

After reviewing the current status of the species, the environmental baseline, climate change, cumulative effects in the action area, and the effects of the continued operation of the seven fisheries under their respective FMPs over the next ten years, it is our biological opinion that the proposed action may adversely affect, but is not likely to jeopardize, the continued existence of North Atlantic right whales, humpback whales, fin whales, and sei whales, or loggerhead (specifically, the NWA DPS), leatherback, Kemp's ridley, and green sea turtles, any of the five DPSs of Atlantic sturgeon, or GOM DPS Atlantic salmon. It is also our biological opinion that the proposed action is not likely to adversely affect hawksbill sea turtles, shortnose sturgeon, smalltooth sawfish DPS, *Acroporid* corals, Johnson's seagrass, sperm whales, blue whales, designated critical habitat for right whales in the Northwest Atlantic, or designated critical habitat for GOM DPS Atlantic salmon.

## 11.0 Incidental Take Statement (including RPMs, T&C)

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." Harm is further defined by NMFS to include any act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. "Otherwise lawful activities" are those actions that meet all State and Federal legal requirements except for the prohibition against taking in ESA Section 9 (51 FR 19936, June 3, 1986), which would include any state endangered species laws or regulations. Section 9(g) makes it unlawful for any person "to attempt to commit, solicit another to commit, or cause to be committed, any offense defined [in the ESA.]" 16 U.S.C. 1538(g). A "person" is defined in part as any entity subject to the jurisdiction of the United States, including an individual, corporation, officer, employee, department or instrument of the Federal government (see 16 U.S.C. 1532(13)). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not the purpose of carrying out an otherwise lawful activity is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement. In issuing ITSs, NMFS takes no position on whether an action is an "otherwise lawful activity."

The prohibitions against incidental take are currently in effect for all four species of sea turtles, endangered whales, the GOM DPS of Atlantic salmon, and all DPSs of Atlantic sturgeon that are listed as endangered. There are currently no ESA-prohibitions for the GOM DPS of Atlantic sturgeon that is listed as threatened. Prohibitions for the GOM DPS of Atlantic sturgeon were proposed in a Federal Register notice published on June 10, 2011 (76 FR 34023). Final action on the proposed rule is pending. The ITS of this Opinion includes the GOM DPS of Atlantic sturgeon in the event of a final rulemaking establishing ESA-prohibitions for the GOM DPS. That part of the ITS pertaining to the GOM DPS of Atlantic sturgeon will be in effect only if the final action, as described in a Federal register notice, is to implement ESA-prohibitions for the DPS.

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

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. NMFS recognizes that further efforts among stakeholders are necessary to reduce interactions between authorized federal fisheries and right, humpback, fin, and sei whales in order to achieve the MMPA's goal of insignificant levels of incidental mortality and serious injury of marine mammals approaching a zero mortality and serious injury rate, taking into consideration the economics of the fishing industry, the availability of existing technology, and existing State or regional fishery management plans. NMFS continues to work toward this zero mortality goal of the MMPA through the means identified in the pertinent subsections of section 5.4 above, including continued development and implementation of the ALWTRP with the collaboration of the ALWTRT. Although NMFS has concluded that the seven fisheries are not likely to jeopardize the continued survival or recovery of right, humpback, fin, and sei whales for purposes of ESA section 7, the need for further efforts among stakeholders to reduce whale/fishery interactions and achieve the zero mortality goal of the MMPA is not diminished by this no-jeopardy conclusion.

### 11.1 Anticipated Amount or Extent of Incidental Take of Sea Turtles

Based on the Murray (2009a) and Warden (2011a) reports, incidental capture data from observer reports for the fisheries assessed in this Opinion, entanglement records from the STDN, and the distribution and abundance of sea turtles in the action area, NMFS anticipates that the continued operation of the seven fisheries may result in the incidental take of sea turtles as follows:

- for loggerhead sea turtles from the NWA DPS, NMFS anticipates (a) the annual take of up to 269 individuals over a five-year average in gillnet gear, of which up to 156 per year may be lethal; (b) the annual take of up to 213 individuals over a four-year average in bottom trawl gear, of which up to 71 per year may be lethal; and (c) the annual take of up to one individual in trap/pot gear, which may be lethal or non-lethal;

- for leatherback sea turtles, NMFS anticipates (a) the annual take of up to four individuals in gillnet gear, of which up to three per year may be lethal; (b) the annual take of up to four individuals in bottom trawl gear, of which up to two per year may be lethal; and (c) the annual take of up to four individuals in trap/pot gear, which may be lethal or non-lethal;

- for Kemp's ridley sea turtles, NMFS anticipates the annual take of up to three individuals in gillnet gear, of which up to two per year may be lethal, and the annual take of up to three individuals in bottom trawl gear, of which up to two per year may be lethal; and

- for green sea turtles, NMFS anticipates the annual take of up to four individuals in gillnet gear, of which up to three per year may be lethal, and the annual take of up to three individuals in bottom trawl gear, of which up to two per year may be lethal.

The anticipated level of incidental take of sea turtles for the recreational components of the bluefish, multispecies, and FSB fisheries cannot be estimated at this time.

### 11.2 Anticipated Amount or Extent of Incidental Take of Atlantic Sturgeon

Based on the NEFSC (2011) and ASMFC (2007) reports, incidental capture data from observer reports for the fisheries assessed in this Opinion, and the distribution and abundance of Atlantic sturgeon in the action area, NMFS anticipates that the continued operation of the seven fisheries may result in the incidental take of Atlantic sturgeon as follows:

- for Atlantic sturgeon from the GOM DPS, NMFS anticipates (a) the annual take of up to 137 individuals over a five-year average in gillnet gear, of which up to 17 adult equivalents per year may be lethal; (b) the annual take of up to 148 individuals over a five-year average in bottom trawl gear, of which up to 5 adult equivalents per year may be lethal;

- for Atlantic sturgeon from the NYB DPS, NMFS anticipates (a) the annual take of up to 632 individuals over a five-year average in gillnet gear, of which up to 79 adult equivalents per year may be lethal; (b) the annual take of up to 685 individuals over a five-year average in bottom trawl gear, of which up to 21 adult equivalents per year may be lethal;

- for Atlantic sturgeon from the CB DPS, NMFS anticipates (a) the annual take of up to 162 individuals over a five-year average in gillnet gear, of which up to 21 adult equivalents per year may be lethal; (b) the annual take of up to 175 individuals over a five-year average in bottom trawl gear, of which up to 6 adult equivalents per year may be lethal;

- for Atlantic sturgeon from the Carolina DPS, NMFS anticipates (a) the annual take of up to 25 individuals over a five-year average in gillnet gear, of which up to four adult equivalents per year may be lethal; (b) the annual take of up to 27 individuals over a five-year average in bottom trawl gear, of which up to one adult equivalent per year may be lethal; and

- for Atlantic sturgeon from the SA DPS, NMFS anticipates (a) the annual take of up to 273 individuals over a five-year average in gillnet gear, of which up to 34 adult equivalents per year may be lethal; (b) the annual take of up to 296 individuals over a five-year average in bottom trawl gear, of which up to 9 adult equivalents per year may be lethal.

The anticipated level of incidental take of Atlantic sturgeon for the recreational components of the seven fisheries cannot be estimated at this time.

### 11.3 Anticipated Amount or Extent of Incidental Take of GOM DPS Atlantic Salmon

Based on incidental capture data from observer reports for the fisheries assessed in this Opinion and the distribution and abundance of GOM DPS Atlantic salmon in the action area, NMFS anticipates that the continued operation of the seven batched fisheries may result in the incidental take of Atlantic salmon as follows:

- One GOM DPS Atlantic salmon on average annually in gillnet gear, of which a lethal take may occur once every three years; and

- One GOM DPS Atlantic salmon on average annually in bottom trawl gear, of which a lethal take may occur once every two years.

The anticipated level of incidental take of Atlantic salmon for the recreational components of the seven fisheries cannot be estimated at this time.

## 12.0 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, the five DPSs of Atlantic sturgeon, and the GOM DPS of Altlantic salmon in the seven fisheries assessed in this Opinion:

1. NMFS must ensure that any sea turtles, Atlantic sturgeon, and Atlantic salmon incidentally taken in gears used in these fisheries (*e.g.*, gillnet, bottom trawl, trap/pot, and hook and line gear) are handled in a way as to minimize stress to the animal and increase its survival rate.

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

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

4. NMFS must ensure that monitoring and reporting of any sea turtles, Atlantic sturgeon, and Atlantic salmon encountered in fishing gear utilized in the seven fisheries: (1) detects any adverse effects such as serious injury or mortality; (2) detects whether the anticipated level of take has occurred or been exceeded; and (3) collects necessary data from individual encounters (e.g. photos, length measurements).

### 12.1 Terms and Conditions

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

1. To comply with RPM #1 above, NMFS must ensure that all Federal permit holders in these fisheries possess handling and resuscitation guidelines for sea turtles, Atlantic sturgeon, and Atlantic salmon. For sea turtles, all Federally-permitted fishing vessels should have the handling and resuscitation requirements listed in 50 CFR 223.206(d)(1) and as reproduced in Attachment A. NMFS must also ensure that all Federal permit holders that use gillnet and trap/pot gear possess the Northeast Region STDN Disentanglement Guidelines. For Atlantic sturgeon, NMFS must instruct fishermen and observers to resuscitate any individuals that may appear to be dead or unresponsive by providing a source of running water over the gills.

2. To also comply with RPM #1 above, NMFS must continue to develop and distribute training materials for commercial fishermen in the use of any release equipment and/or handling protocols and guidelines for sea turtles, Atlantic sturgeon, and Atlantic salmon. Such training materials would be able to be brought onboard fishing vessels and accessed upon incidental capture (*e.g.*, a CD that could be used in on-board computer, placard, etc.).

3. To comply with RPM #2 above, NMFS must continue to investigate modifications of gillnet and bottom trawl gear and its effects on sea turtles, Atlantic sturgeon, and Atlantic salmon 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.

4. To comply with RPM #3 above, NMFS must continue to review all data available on the observed/documented take of sea turtles, Atlantic sturgeon, and Atlantic salmon in Atlantic gillnet, bottom trawl, and trap/pot fisheries and other suitable information (*i.e.*, data on observed interactions for other fisheries, vertical line density information, distribution information, or fishery surveys in the area where the seven fisheries operate) 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, Atlantic sturgeon, and Atlantic salmon interactions and/or their impacts.

5. To comply with RPM #4 above, NMFS fisheries observers must continue to monitor the seven fisheries to document and report incidental bycatch of sea turtles, Atlantic sturgeon, and Atlantic salmon. Monthly summaries and an annual report of observed sea turtle takes in New England and Mid-Atlantic fisheries, including trips where species from these seven FMPs are landed, should continue to be provided to the NERO Protected Resources Division. A similar data sharing plan should be developed for Atlantic sturgeon and Atlantic salmon.

6. To also comply with RPM #4 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.

7. To also comply with RPM #4 above, NEFOP must continue to tag and take tissue samples from incidentally captured sea turtles as stipulated under their ESA section 10 permit. The current NEFOP protocols are to tag any sea turtles caught that are larger than 26 centimeters in notch-to-tip carapace length and to collect tissue samples for genetic analysis from any sea turtles caught that are larger than 25 centimeters in notch-to-tip carapace length. The NEFSC shall be the clearinghouse for any genetic samples of sea turtles taken by observers. Observers must also take fin clips from all incidentally captured Atlantic sturgeon and Atlantic salmon and send them to NMFS for genetic analysis. Observers must ensure that fin clips are taken according to the procedures outlined in Appendices B and C, and that they are taken prior to preservation of other fish parts or whole bodies.

8. To also comply with RPM #4 above, NMFS must continue to utilize and implement sea turtle serious injury guidelines 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. New data should be reviewed on an annual basis.

**Justification for Proposed Reasonable and Prudent Measures and Terms and Conditions**

The RPMs, with their implementing terms and conditions, are designed to minimize and monitor the impact of incidental take that might otherwise result from the proposed actions. Specifically, these RPMs and Terms and Conditions will ensure that NMFS monitors the impacts of the proposed actions in a way that allows for the detection, identification, and reporting of all interactions with ESA-listed species. The discussion below explains why each of these RPMs and Terms and Conditions are necessary or appropriate to minimize or monitor the level of incidental take associated with the proposed action. The RPMs and Terms and Conditions involve no more than a minor change to the proposed actions.

RPM #1 and Terms and Conditions #1 and #2 are necessary and appropriate to ensure that any sea turtles or Atlantic sturgeon that survive capture or entanglement in gear are given the maximum probability of remaining alive and not suffering additional injury or subsequent mortality through inappropriate handling. This is only a minor change as following these procedures is not expected to result in an increase in cost or a decrease in the efficiency of the operation of these fisheries.

RPM #2 and Term and Condition #3 are necessary and appropriate because they allow NMFS to design, research, and implement the most advanced gear modifications believed to have the lowest potential of interactions with sea turtles and Atlantic sturgeon. If gear modifications are implemented, rulemaking will be completed in a timely manner in which to minimize any increase in costs or any decrease in efficiency of the fisheries, representing only a minor change to the actions.

RPM #3 and Term and Condition #4 are necessary and appropriate because they allow NMFS to mitigate sea turtle and Atlantic sturgeon associate with environmental conditions or other parameters present in the action area. If regulations are implemented, rulemaking will be done in a manner in which to minimize any increase in costs or any decrease in efficiency of the fisheries, representing only a minor change to the actions.

RPM #4 and Terms and Conditions #5, #6, #7, and #8 are necessary and appropriate to ensure the proper documentation of any interactions with sea turtles and Atlantic sturgeon as well as requiring that these interactions are reported to NMFS in a timely manner with all the necessary information. This is essential for monitoring the level of incidental take associated with these seven fisheries. Compliance with these terms and conditions will allow NMFS to determine if reinitiation of consultation is necessary at the time that take occurs. The data and information collected can be used to refine our current management measures, and is not just a count of dead or injured individuals. This RPM and its Terms and Conditions represent only a minor change as compliance is not expected to result in an increase in cost or a decrease in the efficiency of the fishery operations.

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

**Sea Turtle Monitoring**

NMFS must continue to monitor levels of sea turtle bycatch in the seven fisheries. Observer coverage has been used as the principal means to estimate sea turtle bycatch in the gillnet, trawl, and longline fisheries and to monitor incidental take levels. Entanglement reports have been used as the principal means to estimate sea turtle bycatch in the pot/trap fisheries and to monitor incidental take levels. NMFS must continue to use observer coverage and entanglement reports to monitor sea turtle bycatch in gear that is authorized by the FMPs for the seven fisheries.

For the purposes of monitoring this ITS for the gillnet, trawl, and bottom longline components of the seven fisheries, we will continue to use observer coverage as the primary means of collecting incidental take information, taking into account regional observer coverage levels by gear type. The loggerhead sea turtle take estimates described in this 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 the 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 five years, we will estimate takes in the seven fisheries using appropriate statistical methods. Additionally, on an annual basis we will review observed takes of loggerhead turtles to consider trends in takes and look for patterns and changes in take levels. For sea turtle species other than loggerheads, we will use all available information (*e.g.*, observed takes, changes in fishing effort, etc.) to assess if the annual incidental take level in this Opinion has been exceeded.

For the purposes of monitoring the ITS in regards to sea turtles that are known to be entangled in pot/trap gear, NMFS will continue to use STDN data as the primary means of collecting incidental take information. NMFS will assess takes annually in the seven fisheries using all available and up to date STDN entanglement data. Using these data, NMFS will determine if the annual incidental take level in this Opinion has been met or exceeded.

**Atlantic Sturgeon Monitoring**
NMFS must monitor levels of Atlantic sturgeon bycatch in the seven fisheries. Observer coverage has been used as the principal means to estimate Atlantic sturgeon bycatch in gillnet and trawl fisheries, and will be used to monitor incidental take levels in gear that is authorized by the FMPs for the seven fisheries.

For the purposes of monitoring this ITS for the sink gillnet and trawl components of the seven fisheries, we will continue to use observer coverage as the primary means of collecting incidental take information. As the 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 five years, we will re-estimate takes in the seven fisheries using appropriate statistical methods. For the five Atlantic sturgeon DPSs, we will use all available information (*e.g.*, observed takes, changes in fishing effort, etc.) to assess if the annual incidental take level in this Opinion has been exceeded.

**GOM DPS Atlantic Salmon Monitoring**
NMFS must monitor levels of Atlantic salmon bycatch in the seven fisheries. Observer coverage has been used as the principal means to estimate Atlantic salmon bycatch in the gillnet and trawl fisheries, and will be used to monitor incidental take levels in gear that is authorized by the FMPs for the seven fisheries.

For the purposes of monitoring this ITS for the sink gillnet and trawl components of the seven fisheries, we will continue to use observer coverage as the primary means of collecting incidental take information. For the GOM DPS Atlantic salmon, we will use all available information (*e.g.*, observed takes, changes in fishing effort, etc.) to assess if the annual incidental take level in this Opinion has been exceeded.

**Large Whale Monitoring**
NMFS will continue to monitor levels of large whale entanglement in the seven fisheries. Serious injury determinations and stock assessment reports have been used as the principal means to estimate the large whale entanglement rate in the seven fisheries and to monitor SI/M levels. NMFS has recently developed a monitoring strategy for the ALWTRP and will produce an annual report stating the most up-to-date SI/M five year rolling average. To provide the most up-to-date rolling average possible, the five-year average will consist of the most recently available year's data from the annual SI/M report averaged with the previous 4 years of data obtained from the U.S. Atlantic and Gulf of Mexico Marine Mammal SAR. Analyzing the data in this way will reduce the two year lag associated with using SAR estimates alone by one year.

For the purposes of monitoring large whale SI/M, NMFS will use the serious injury determination reports, SARs, and the ALWTRP monitoring reports to collect entanglement information. NMFS will re-examine SI/M annually in the seven fisheries. Using these data, NMFS will determine if the annual SI/M is significantly different than what was evaluated in this Opinion.

## 13.0 Conservation Recommendations

In addition to section 7(a)(2), which requires agencies to ensure that proposed actions are not likely 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 ESA-listed species conservation:

1. NMFS should continue to collect and analyze biological samples from sea turtles incidentally taken in trawl, gillnet, bottom longline, and pot/trap fishing gear to determine the nesting origin of sea turtles taken in the gear types used in the seven fisheries in order to better assess the effects of these fisheries 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 fishing gears used in the seven fisheries 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 and properly transferred to appropriate personnel for examination. The protocol should also 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 continue to collect and analyze biological samples from Atlantic sturgeon incidentally taken in fishing gear to determine the DPS origin of Atlantic sturgeon taken in the gear types used in the seven batched fisheries in order to better assess the effects of these fisheries on each DPS 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 Atlantic sturgeon.

4. NMFS should collect and analyze biological samples from Atlantic salmon incidentally taken in fishing gear to determine possible GOM DPS origin of Atlantic salmon taken in the gear types used in the seven batched fisheries in order to better assess the effects of these fisheries on GOM DPS of Atlantic salmon and address those effects accordingly. NMFS should create its policies/protocols for the processing of genetics samples to maximize the efficiency for obtaining results of genetic samples taken from all incidentally taken Atlantic salmon.

5. NMFS should work with 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.

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

7. 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.

8. NMFS should continue to undertake and support aerial surveys, passive acoustic monitoring, and the Sighting Advisory System.

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

10. 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.

11. 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 ESA-listed species.

## 14.0 Reinitiating Consultation

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

In addition, re-initiation will be required if NMFS determines that in any given calendar year following the release of this biological opinion one or more of the following has occurred as a result of U.S. federal fisheries and in gear used or possibly used under the seven batched FMPs: (1) more than three mortalities or serious injuries of North Atlantic right whales; or (2) more than eight mortalities or serious injuries of humpback whales; or (3) more than three mortalities or serious injuries of fin whales; and/or (4) more than two mortalities or serious injuries of sei whales.

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# 16.0 APPENDIX A: Scaling the Influence of Anthropogenic Mortality Reduction on Recovery Prospects of North Atlantic Right Whales

# SCALING THE INFLUENCE OF ANTHROPOGENIC MORTALITY REDUCTION ON RECOVERY PROSPECTS OF NORTH ATLANTIC RIGHT WHALES

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**Abstract**

Understanding the influence of reducing human-caused mortality on the magnitude of changes to the growth in small populations of long-lived, slowly reproducing organisms would help managers gauge the value of conservation measures aimed at recovering the species. The critically endangered North Atlantic right whale (*Eubalaena glacialis* Müller 1776) represents an interesting case study toward that end because considerable effort has been and continues to be spent in targeting conservation measures at specific mortality causes. I built simulation models of annual population change that re-sampled from observed calving records and estimates of survival rates for the period 1980-2007. I used these to assess the influence that various reductions in anthropogenic mortality might have on status quo simulations. The reductions investigated included a simple scenario of saving one adult female per annum from mortality as well as the per capita equivalent of saving 1 animal per year over a 100 year evaluation horizon. Status quo, simple and per capita simulations produced median overall growth rates of 1.06, 1.43 and 1.51% respectively. Because projections frequently showed considerable growth, per capita reductions resulted in adding back about 200 adult females over the 100 year time frame. These results indicated that, if growth under status quo conditions is about 1%,

then saving 1 adult female per year or even its per capita equivalent, although modestly improving right whale population viability, would unlikely result in a population rate of increase exceeding the 2% growth threshold for 35 years as required by the right whale recovery plan. If actual current growth is nearly 2% as recently reported, then saving 1 adult female per year would likely meet recovery goals.

**Key-words:** by-catch, *Eubalaena glacialis*, ship strikes, survival, viability analysis

## 1. Introduction

Conservation biology is rife with examples of conservation measures having substantial influence on population growth of small populations as well as the crippling effects of environmental variability and catastrophes reducing population growth (Hunter, 1996). However, many long-lived animals have evolutionary stable strategies that place a premium on survival, often allowing adult individuals to weather the storms of low resources by forgoing reproduction. Consequently, continued small amounts of anthropogenic mortality can have a profound effect on the long-term viability of such populations. Additionally, conservation measures aimed at increasing survival might be expected to produce only modest improvements to growth in cases where adult survival was already likely very high and fecundity is inherently low. Therefore, managers need some realistic expectations of the scale of the impacts of their conservation actions on such populations.

North Atlantic right whales are usually described as critically endangered (Kraus et al., 2001) and have received considerable attention from U.S. National Marine Fisheries Service (NMFS) because various human activities (e.g. commercial fishing and shipping) can cause mortality to large whales (Knowlton and Kraus 2001, Glass et al., 2009). The latest published Recovery Plan for the species (National Marine Fisheries Service, 2005) states that entanglements in fishing gear and collisions with vessels have been the primary cause for a lack of recovery of the North Atlantic right whale population. Evaluations of injury records found that, during the period 1998-2007, known deaths or serious or serious injuries to a right whales due to entanglement and collisions with vessels averaged1 and 2 per year respectively (Cole et al., 2005, Glass et al., 2009, NEFSC unpubl. data). Various published and unpublished right whale population assessments have been produced since 1999. Among these assessments are those of Caswell et al. (1999) and Fugiwara and Caswell (2001), both of which concluded that the right whale population was in decline. The latter study conjectured that saving 1 adult female per year would reverse that decline.

Population simulation studies, generally referred to as Population Viability Analyses (PVAs), are a common tool used to evaluate conservation strategy impacts on wildlife populations, especially for species threatened by or in danger of extinction (Morris and Doak, 2002). PVA models are developed using general knowledge of the dynamics of a population combined with available observational data and estimates of demographic parameters. For North Atlantic right whales, considerable life history data exist for creating a robust population projection model (see Kraus and Rolland, 2007). The ability to identify individual whales at an early age coupled with annual surveys of most whale habitats for more than 25 years has generated an extensive individual sightings database of most animals in the population. From these data, survival and fecundity rates can be estimated and followed through time with moderate precision, including some age-specific mortality information. Herein, I produce estimates of right whale survival rates. I use these together with observed annual calving rates and some basic demographic assumptions to develop a stochastic population projection model. I then modify this model to evaluate the effects that saving 1-3 whales annually would have on population projections and the rate of population recovery.

## 2. Methods

To develop a stochastic simulation model with which to evaluate the effect of reduced anthropogenic mortality on the projected dynamics of the North Atlantic right whale population, I used information developed from sightings records of individuals collected over 26 years. Specifically, I used re-sightings histories of known individuals to estimate survival rates, and used counts of calves to estimate fecundity rates. I re-sampled these rates in a cumulative, stage-based population simulator.

### 2.1 Survival Rates

I acquired a listing of 32,591 sightings of cataloged individual North Atlantic right whales extracted on 30 May 2007 from a database curated by the New England Aquarium (NEAq, Boston, Massachusetts, USA). Identifications of individual whales were provided by NEAq personnel and based primarily on photo-identification using natural markings (Kraus et al., 1986, Hamilton et al., 2007) and supplemented with genetic markers (Frasier et al., 2007). Recapture histories of individuals were accumulated during various survey efforts conducted in 5 principal geographic regions along the Atlantic coast of North America (Brown, et al., 2007). Although several individual whales were identified and sighted multiple times prior to 1980, annual data acquisitions since 1980 are more abundant, acquired more systematically and have received the most analysis from other researchers. Spatial coverage has varied considerably since 1980, but for each year from 1980 to 2007 sightings data were available nearly year round from across much of the known range of the species. I collated multiple daily sightings of the same individual into a single sighting event and examined the distribution of sighting events within and among years and among habitats.

Due to the diligence required to process the large number images taken during multiple surveys searching for North Atlantic right whales, a significant lag currently exists before the sighting records of each individual whale are resolved and available for review. An appreciable number of photographic captures taken during 1 December 2005-30 May 2007 had not been fully processed by 30 May 2007 when I acquired the sightings data. I therefore selected data from 1 December 1979-30 November 2005 that, although not complete for all habitats sampled during 2004-5, would adequately allow for estimation of 2004 survival rate in the completely time varying model. [In the complete time-specific survival and capture probability model the last survival interval (i.e., 2004-2005) is not identifiable (Williams et al., 2001)]. For the years 1980-2005, the 32,591 records (reduced to one per day) comprised 498 live individual whales. Of these whales, 394 and 206 were of known sex and age, respectively. There were 188 females and 206 males. From these data, I selected two overlapping subsets: one for estimation of annual survival rates in a modified CJS model (Lebreton et al., 1992) and one for use in a composite dynamic model of age specific mortality.

2.2.1 The CJS model.

Classical open-population mark-recapture models make assumptions of homogeneity within groups to which estimated values apply (Williams et al., 2001), and most long-lived mammals show variation in survival rates by age (Caughley, 1966). Because I was concerned that lower survival rates of the youngest animals would add appreciable heterogeneity to the data used in a CJS model, I excluded all capture records for animals known to be <4 years of age, leaving 28 197 daily sightings of 469 individual right whales distributed among geographic areas.

A priori, I suspected different capture and survival probabilities for each sex. In the southeast US (SEUS), recaptures are primarily of calving females, a few of which are not seen elsewhere, their calves and young animals. Behaviors such as participation in surface active groups (Parks et al., 2007) are male dominated, highly visible and offer good photographic opportunities of the participants. In addition, some evidence suggests that sighting probabilities may differ among age groups and because of differential use of survey areas (Brown et al., 2001). Adding to possible differences in 'catchability', several aspects of sex-related behavior (calving, calf protection, male competitions for mates) expose sexes to different hazards and could result in different mortality schedules. For example, females normally migrate and endure an extended fast while calving and through the first few months of their calf's life (Kraus et al., 2007). Therefore, I considered 3 sex groups when estimating annual capture and survival rates: male, female and unknown sex. The unknown sex group was problematic in that it was composed of individuals that provided fewer opportunities to determine their sex (i.e., shorter lived, younger, or seen infrequently), but failure to consider them biases the estimated rates (Nichols et al., 2004).

Prior to computing survival estimates, I used program U-Care (v.2.2, Choquet, et al., 2005) to evaluate the structure of annual sighting histories, grouped by sex, by testing several catchability hypotheses. In particular, I was interested in evidence for transients and evidence for capture proneness.

I used program MARK to calculate estimates in various CJS models (White and Burnham, 1999). Capture histories were built by compressing sighting records of individual whales during a year (often multiple sightings of the same individual in multiple geographic areas on nearly any day of the year) into a single binary observation (seen or not seen). A year was defined as December 1 – November 30, except that the few November sightings of whales made in the SEUS were attributed to the subsequent year (which promotes a more realistic categorization because in November these individuals have just completed a southerly fall migration). I thought that considerable information about the catchability of individuals was contained in the within-year capture histories. Therefore, I constructed an individual whale covariate which was the average adjusted rank of number of recaptures within years in which each individual whale was seen. Prior to averaging, I adjusted the set of ranks for each year by dividing by the total number of individual whales seen in that year, thus placing all years on the same basis (0-1). I was specifically interested in annually varying survival rates because such variability increases fluctuations in annual population size and therefore increases the potential for extinction (Morris and Doak, 2002). Accordingly, I was only interested in CJS models with one of the following survival parameterizations: constant survival over time, a linear trend in survival over time and freely time-varying survival. The effect of sex on survival was allowed to be zero, additive or independent of annual effects. I used AIC to compare various combinations of parameters and selected the best fitting models to provide status quo survival rates for population simulations.

2.1.2 Adjustments to survival rates

As noted earlier, only 41% of the animals in the database were of known age. Due to variations in sampling effort and calf production, initial sightings of these animals were distributed unevenly among years. As a substitute for creating an age-specific mark-recapture model that incorporated the uncertainty of unknown ages, I used all known age animals (n=206) to create a composite (pooled across years and sexes) table to estimate age-specific mortality rates of ages 0-5+ (Caughley, 1977), a period during which differences in survival by sex are not as likely as among animals of reproductive age. I tabulated numbers of animals at risk (known to be alive) by age class and those known to be alive in the subsequent year. I fitted a binomial model with age as a categorical predictor to these data. I assumed that calf data only represented ½ a year and squared the predicted survival rate. I used these estimated survival rates divided by the estimated survival for animals older than 4 to create odds ratios. These ratios were used to adjust the annual survival rates for age groups 0-4 in the population simulations described below.

## 2.2 Fecundity

Maintained along with the sightings histories of individuals are annual calf production data (Kraus et al., 2007). To use the annual calf counts of 1980-2007, I adjusted them to per capita rates based on recent right whale population trends. Waring et al., (2009) reported that the minimum number of right whales alive during 1990-2004 increased at an annual rate of about 5 animals per year. Starting with 350, the minimum number alive in 2006, and adjusting each prior year by 5 animals downwards, I divided the calf counts for 2006, 2005, 2004, …, 1980 by adjusted population sizes of 350, 345, 340, …, 215, respectively to yield a set of predicted per capita reproduction values from which to sample in the population simulations. Predicted per capita rates were each multiplied by 345 (starting population size for all simulations, see section 2.3) to produce a set of adjusted calf counts from which to sample.

## 2.3 Status Quo Population Simulation

I used sampling with replacement to select annual survival and fecundity rates and applied these to stage-structured populations as binomial processes in 100 steps to simulate status quo population changes over a 100-year time frame. Models tracked six male (age 0-5+) and 14 female (age 0-13+) stages (Fig. 1). The basics of the simulation entailed three parts: establishing the starting population structure, stepping this structure through time by generating a new cohort, and applying annual survival functions to all stages. This process was then repeated, all the while plotting or collecting statistics for summary.

### 2.3.1 Starting values

The initial total population size for all projections was 345, but the initial distributions of these 345 animals across sex and age groups varied. To establish an initial sex and age distribution, I first established the initial total number of males at start (T.males), as binomial (n=345, p=0.5). Then, I produced 13 cohorts by equal probability sampling with replacement 13 times from the distribution of adjusted calf counts. I assigned a binomial (n=cohort size, p=0.5) random number of these to be males. I reduced male counts for each age class 0-4 by applying the product of the appropriate annual age-specific survival rates in a binomial selection process, and let T.males minus the sum of the generated age classes 0-4. I used a similar process to reduce the number of female calves for each cohort to numbers at age by applying the appropriate period specific survival rates in a binomial survival process. The 13+ female age class was the 345 population size minus the sum of all other cohorts (T.males + sum of the generated female age classes 0-12). To reflect the possible difference in the number of males and females found in the catalog data, I further reduced the size of the initial 13+ female age class by choosing binomial (n, p=0.9). Those not selected were added to the male 5+ class to keep the initial population size at 345.

A set of fecundity values was also established at the start of each projection. This was simply a randomly selected adjusted calf count divided by the sum of the initial female 10 and 13+ stage class (Fig. 1).

2.3.2 Time steps

Each time step required application of stage-specific survival functions to generate the population size and structure for the next step, followed by the production of a new cohort (age class 0) based on the established population structure. This was accomplished by incorporating certain life history characteristics of right whales into the model. In particular, I assumed that female whales were sexually mature at 10 and most (90%) calved at that stage. Further, individual females were not allowed to calve for 2 years following a successful calving (see Kraus et al., 2007 for support for these assumptions). I also used evidence that females in calving years incurred a small mortality penalty (Fujiwara and Caswell, 2001; NMFS, Woods Hole, MA unpublished data) to adjust postpartum female mortality.

Except for stage 0, 11 and 13+ females, sizes of each class at time t+1 was assumed to be the result of a binomial(n(i), p(i)) process. Here, n(i) and p(i) were, respectively, the number at time t and stage-specific mortality of the ith stage. Stage-specific survival rates over the interval of time from year t to t+1 were derived by randomly selecting from amongst the sets of estimated sex-specific survival rates described previously and applying the stage-specific odds ratios to that selection. Thus, the effect of age on survival was assumed to be multiplicative to annual fluctuations in overall survival. The size of age class 0 was derived from a reproductive function described below. The number of stage 11 females was the number of stage 10 females that produced calves and survived plus stage 13 females that produced calves and survived. The number of stage 13 females at time t+1 was the sum of non-calving 10 and non-calving stage 13 females that survived. Survival rate for calving females was set to 0.9 times the selected overall survival rate.

Annual reproduction was determined as the sum of separate binomial processes using the counts of stage 10 and stage 13 females with probabilities of calving equal to 0.9 and a randomly selected fecundity rate, respectively. The fecundity rate was a random pick from the adjusted calf production values divided by the size of the initial number of stage 13 females. The number of females that calved at time t+1 was set equal to the number of calves generated for time t+1.

2.3.3 Summary statistics

For the status quo simulation and for each of 5 sets of mortality reductions described below, I replicated 100-year population projections 1000 times. For each time step within a projection, I calculated the total population size and randomly selected and plotted 25% of projections. For each projection, I calculated population growth statistics, lambda and lambda.36, as:

$$\text{lambda} = \exp((\log(\text{N.final})-\log(345))/100)$$

$$\text{lambda.36} = \exp((\log(\text{N.36})-\log(345))/36),$$

where N.final and N.36 were total population size at step 100 and 36, respectively. Population growth at year 36 was of interest because it coincided with a 35 year time horizon over which population growth must equal or exceed 2% in order to declare the species recovered (National Marine Fisheries Service, 2005). I calculated the median values for lambda and lambda.36, plotted histograms to show the observed variability in predicted population change and produced cumulative distributions to evaluate probabilities of projections exceeding the recovery thresholds. I also counted the number of projections that declined by the end of the projection period (i.e., N.final<345), the number of projections in which the population went extinct (N.female<1), and the number of projections in which the population declined to a quasi-extinction threshold of less than 25 reproductive females (i.e., N[class 10-13 females]< 25).

2.4 Modifications to Status Quo

I wanted to estimate the possible effect that reducing mortality, especially adult female mortality, would have relative to status quo projections, and then use that effect to assess the potential for recovery of the North Atlantic right whales population. Specifically, I examined 3 levels of adding back animals to the population (a single animal, an adult female, 3 animals) under 2 scenarios: simple and per capita. In all instances, the status quo model was modified to affect the increase only in cases where the equivalent level of mortalities occurred. For example, to add back adult females, at least 1 female from any stage 10-13 must have died during that time step. For simple reductions in mortality, I added back 1 or more adult females during years in which at least 1 adult female was projected to die under status quo, so that the number of additions never exceeded the number of deaths and averaged 1 per year over the entire projection. For the model adding back any stage class, the stage added back was selected randomly from a multinomial distribution with probability equal to the proportion dying among all deaths. In the case of per capita reductions, I first calculated the per capita cause-specific mortality rate equivalent to 1 adult female for the starting age structure of each projection. For each year, I added back the lesser of the number projected to die among adult females and a random binomial number with class size as the number of trials and the calculated per capita cause-specific mortality as the probability of success. I included a model in which the per capita reductions were equivalent to overall mortality of 3 animals out of a population of 345 (0.87%). Mortality rates equivalent to one and three deaths for the starting population size of 345 were selected for study because they coincided respectively with the numbers of detected entanglement-related and total human caused mortalities report by NMFS in a recent report (Waring et al., 2009).

For both status quo and modified simulations, I calculated per capita mortality rates and averaged those within and among projections. For modified simulations, I averaged the number of whales added back each year. All simulations were developed and performed using the R mathematical base package (R Development Core Team, 2009) with random seeds to start all random number generation processes.

## 3. Results

After compressing daily capture histories to a binary outcome (seen or not seen) each year (defined as December 1 – November 30) and on the basis of Goodness-Of-Fit (GOF) tests, I found no evidence for transients within capture histories of North Atlantic right whales ($P>0.2$). Thus, despite some whales being seen relatively infrequently, individual capture heterogeneity did not manifest itself in a way that would be expected to depress apparent survival rate estimates as would the presence of transients within a mark-recapture data set (Pradel et al., 1997). Conversely, these data showed ample evidence of a 'trap happiness' among individual whales ($P<0.05$). It seemed unlikely that this is a true behavioral response, but is a consequence of joint whale and researcher geographic (habitat) fidelity. That is, many individuals (whales and photo-taking humans) return to the same core areas each year with the result being many individual whales with recapture rates higher than the overall capture rates. Fortunately, behavioral response has little to no effect on estimates of survival in CJS models because survival is a function of the marked portion of the population (Nichols et al., 1984).

### 3.1 Survival Rate Estimates

Among the models examined, two provided nearly equal fits to the data based on AIC. Both included additive sex effects to survival and capture probabilities and incorporated an individual catchability coefficient in the estimation of capture probabilities. One model used a linear (in the logistic) decreasing trend to fit survival rates while the alternative was unconstrained time varying survival rates. I selected the latter model to provide survival estimates for simulation because the additional variance among survival rates would induce greater fluctuations in population projections and therefore more uncertainty among outcomes (Morris and Doak, 2002). For the period 1980-2005, estimated survival rates from sighting histories of all animals known to be not less than 4 years old ranged between 0.926 and 1.0 for females and 0.945 and 1.0 for males (Fig. 2). Because the observed variability among these estimates is due to the combined effects of sampling variation and biological variation any simulation model that samples from them directly is likely hypervariable relative to the biological processes themselves (White et al., 2002).

Based on a generalized linear model of age-specific survival (composite dynamic table), adjustments to survival rates to accommodate lower survival of young animals were warranted (Probability of no age variation <0.001). Estimated survival rates for ages 0,1,2,3, 4, and 5+ animals were 85.0, 91.6, 91.0, 88.2, 92.0 and 97.0%, respectively. Therefore, odds-ratios used to reduce the survival rates for animals aged 0-4 years were 0.876, 0.944, 0.938, 0.909, and 0.948.

### 3.2 Fecundity Values

Calf counts, available from NEAq, were not used directly in simulation models, but observed calf counts were adjusted to per capita reproduction rates (Table 1). I used 2 assumptions: that a total population of 345 produced the number of calves observed in 2005 and that the population had grown at a rate of 5 animals per year between 1980 and 2007. Observed calf counts were thus adjusted by multiplying the per capita reproduction rates by the starting population size of each simulation, 345 (Fig. 3).

### 3.3 Population Simulations

#### 3.3.1 Status quo projections

Status quo projections produced a very low likelihood for extinction in this population (Fig. 4 A). No extinctions or quasi-extinctions were observed. Only 2 of 1000 projections ended with smaller total population size than they started, and those were just marginally smaller. Median growth rate among status quo projections was 1.3% over the entire 100-year period (Fig. 5 A), and 1.38% for the first 35 years (Table 2). The overall per capita mortality rate averaged across years and projections was 3.59%. Only 8.6 and 4.1% of status quo simulations achieved 2% or better overall growth for the first 35 years after start and entire 100 year duration of projections, respectively (Fig. 6).

#### 3.3.2 Reductions in mortality

The distributions of the overall growth statistics lambda and lambda.36, showed only modest increases following modifications to survival processes that reduced annual mortality equivalent to 1 whale relative to status quo mortality schedules (Fig 4 - 6). The smallest gains in lambda were produced by adding back one whale per year, and resulted in no observed projection that had a final population size less than the starting population size of 345. Much more substantial increases in overall growth were observed in the simulation scenario that modeled elimination of the per capita equivalent of the average detected human caused mortality of 3 animals per year. Of the five reduction scenarios modeled, adding back the per capita equivalent of 3 in 345 deaths was the only scenario that produced at least 50% of projections that surpassed the 2% growth threshold for recovery.

## 4. Discussion

In many respects, the dynamical aspects of the North Atlantic right whale population are typical among large bodied, long-lived mammals not in substantial decline, that is, very high survival and low fecundity. Although the North Atlantic right whale population is often described as critically endangered, it currently fails to meet the standards for that categorization as defined by the International Union for Conservation of Nature and Natural Resources (Reilly et al., 2008). Nonetheless, by all accounts, the population is exceedingly small (~400 individuals) and its growth is likely constrained by human-caused mortality. Still, the population has modestly increased in size (averaging a net gain of 5 animals per year) over at least 15 years (Waring, et al., 2009). The observed increases contradict several relatively recent population assessments which had pronounced the species doomed over the next 300 years (e.g., Caswell et al., 1999). However, the failures of the earlier forecasts to predict recent population trends do not totally discredit those assessments. Indeed, those assessments highlight the potential for small differences in estimated demographic values to alter the predicted fates of any population of long-lived individuals. One possible cause for the incorrect forecasts is that the true values of recruitment and survival for the population were very close to those that would produce a sustained decline.

My work indicates a tenuous future for North Atlantic right whales while better reflecting the observed growth in the population. Although my simulations better track the recent growth in the right whale populations, several aspects of the models are open to criticism. First, the population projection models are hypervariable. In this instance, hypervariability was by design. Specifically, I chose to include a more variable set of survival estimates than those produced by a competing model of similar statistical support. As well, I did not attempt to remove an estimate of sampling variation from survival estimates so as to capture the full range of realized annual survival rates possibly observed in this population over the past 25 years. Therefore, the more extreme (nearly flat or rapidly growing) population projections are probably less likely than their frequency of occurrence among the simulated outcomes. Because increased interannual variability in survival rates produces a reduction in population growth over extended periods, the increased variability may have depressed the median level of population increase compared to that observed in the right whale population over in recent years. These simulations included no consideration of density dependence mechanisms, following the recommendation of Morris and Doak (2002) who stated that if no data exist to support and define density dependence, then it should be excluded from consideration.

Of course, any population projection model is only as robust as the data on which it is based. The principal data here included observed numbers of calves. Because survey intensity increased substantially in the late 1980s to ensure more complete calf counts, early values included here and re-sampled with equal probability with latter calf counts may have slightly reduced the potential for population increases the projections. However, the overall trend of increasing calf counts (Kraus et al., 2007) seems to parallel a gradual increase in overall population size, albeit with significant variability.

The other important demographic parameters included in these projection models were estimated from re-sightings histories. CJS models have their own set of data assumptions (Lebreton et al., 2002), and several aspects of the right whale sightings data are not typical of mark-recapture data. Specifically, right whales are sighted during almost every month of the year which makes survival intervals somewhat fuzzy (Smith and Anderson, 1987, Williams et al., 2002). North Atlantic right whales are also well known for differential catchability stemming from different use patterns among the geographic areas regularly surveyed for their presence (Brown et al., 2001). I developed a new calculation of individual catchability coefficients that successfully compensated for much of the individual capture heterogeneity observed in this population. Because coefficients were based on within year sightings, whales observed for many years had many more data to estimate this coefficient. Inclusion of the individual catchability coefficient greatly improved CJS model fits, but considerable capture heterogeneity may still exist. Selecting one set of estimates (full time varying) over those from a competing model (linear decline) which shared similar support is not optimal relative to model averaged estimates and does not recognize the uncertainty of the selection process in the precision of the estimated parameters (Burnham and Anderson, 2002). However, the goal here was not to present or use the precision of the estimated survival rates from the CJS models. Rather, estimates were selected to provide possible process values for stochastic simulations of population change which would encompass the true range of values. I selected the more variable set of estimates as this would generate increased variability within projections; the use of model-averaged estimates would have reduced this variability. Using a set of estimates more likely to encompass the range of natural variability in survival rates was a reasonable approach for accepting the increased frequency of extreme values of survival rates. Because the same basic set of survival estimates were used under all scenarios, using the most variable set provided a conservative test for detecting different population growth patterns under these scenarios.

The results from simulation provide a possible view of the value of reduced human-caused mortality to the slow population growth rates of North Atlantic right whales. In absolute terms, an increase in median growth rate from ~1.3 to 1.6 or 1.8% appears quite small. However, the scale of the increase is still likely to be highly significant. Saving one adult female each year was significantly more effective at improving growth than saving one animal chosen at random from any stage. Furthermore, policies that reduce the effects of mortality factors which take more whales when more whales are present, such as interactions with fishing gear or vessels, will be more effective than measures that save a fixed number of whales each year, such as disentanglement efforts. Whales dying from human-related causes are almost surely undercounted. But if only the detected levels of human-caused mortality due from vessel collisions and gear entanglement (about 3 per year) were removed, my simulations indicate the North Atlantic right whale population's growth rate would likely exceed recovery thresholds for population growth. If the current true population is close to 2% as reported in Waring et al. (2009), then reduction of human-caused mortalities of the kind and scale simulated in this study would almost surely generate a population growth rate above the recovery threshold.

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**Table 1**. Counts of unique calves detected for the western North Atlantic right whale population (Courtesy of Right Whale Consortium Data Base curated by New England Aquarium, Boston, Massachusetts, USA). Included are the assumed population sizes producing those calves and associated per capita calving rates.

| YEAR | CALF COUNT | ASSUMED N | PER CAPITA RATE |
|---|---|---|---|
| 1980 | 6 | 220 | 0.0273 |
| 1981 | 8 | 225 | 0.0356 |
| 1982 | 11 | 230 | 0.0478 |
| 1983 | 9 | 235 | 0.0383 |
| 1984 | 12 | 240 | 0.0500 |
| 1985 | 11 | 245 | 0.0449 |
| 1986 | 13 | 250 | 0.0520 |
| 1987 | 11 | 255 | 0.0431 |
| 1988 | 7 | 260 | 0.0269 |
| 1989 | 16 | 265 | 0.0604 |
| 1990 | 12 | 270 | 0.0444 |
| 1991 | 17 | 275 | 0.0618 |
| 1992 | 12 | 280 | 0.0429 |
| 1993 | 6 | 285 | 0.0211 |
| 1994 | 9 | 290 | 0.0310 |
| 1995 | 7 | 295 | 0.0237 |
| 1996 | 21 | 300 | 0.0700 |

| | | | |
|---|---|---|---|
| 1997 | 5 | 305 | 0.0164 |
| 1998 | 4 | 310 | 0.0129 |
| 1999 | 1 | 315 | 0.0032 |
| 2000 | 31 | 320 | 0.0969 |
| 2001 | 21 | 325 | 0.0646 |
| 2002 | 19 | 330 | 0.0576 |
| 2003 | 16 | 335 | 0.0478 |
| 2004 | 28 | 340 | 0.0824 |
| 2005 | 19 | 345 | 0.0551 |
| 2006 | 22 | 350 | 0.0629 |

**Table 2.** Statistics resulting from 6 PVA models of North Atlantic right whales under different human-caused mortality schedules.

| | MODEL | | | | | |
|---|---|---|---|---|---|---|
| | STATUS QUO | SIMPLE CR[2] | SIMPLE AF[3] | PER CAP CR[4] | PER CAP AF[5] | PER CAP 3[6] |
| Mean mortality reduction[1] | 0 | 100.0 | 99.2 | 251 | 254 | 1063 |
| Median $\Lambda$ (100) | 1.0130 | 1.0150 | 1.0160 | 1.0157 | 1.0180 | 1.0209 |
| Median $\Lambda$ (35) | 1.0138 | 1.0156 | 1.0170 | 1.0162 | 1.0180 | 1.0213 |
| Mean Mortality Rate | 0.0359 | 0.0346 | 0.0353 | 0.0332 | 0.0345 | 0.0280 |
| Projection above 2% Growth (100) | 41 | 61 | 120 | 122 | 280 | 624 |
| Projection above 2% Growth (35) | 86 | 147 | 246 | 173 | 304 | 643 |

[1]Average number of animals saved relative to status quo conditions during 1000, 100-year simulations.
[2]One animal that would have died under status quo conditions added back to a single class chosen with probability equal to fraction dying.
[3]Adding back an average of 1 adult female per year to those years during which at least 1 was simulated to die under status quo conditions.
[4]Adding back the per capita equivalent of 1 animal at initial conditions (1/345)
[5]Adding back the per capita equivalent of 1 adult female at initial conditions (1/number of adult females)
[6]Adding back the per capita equivalent of 3 animals at initial conditions (3/345) to classes chosen with probability equal to fraction dying.

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**Fig. 1.** Schematic diagram of life stages and possible transitions included in viability models designed to examine effects of reduced mortality on prospects for recovering the North Atlantic right whale population.

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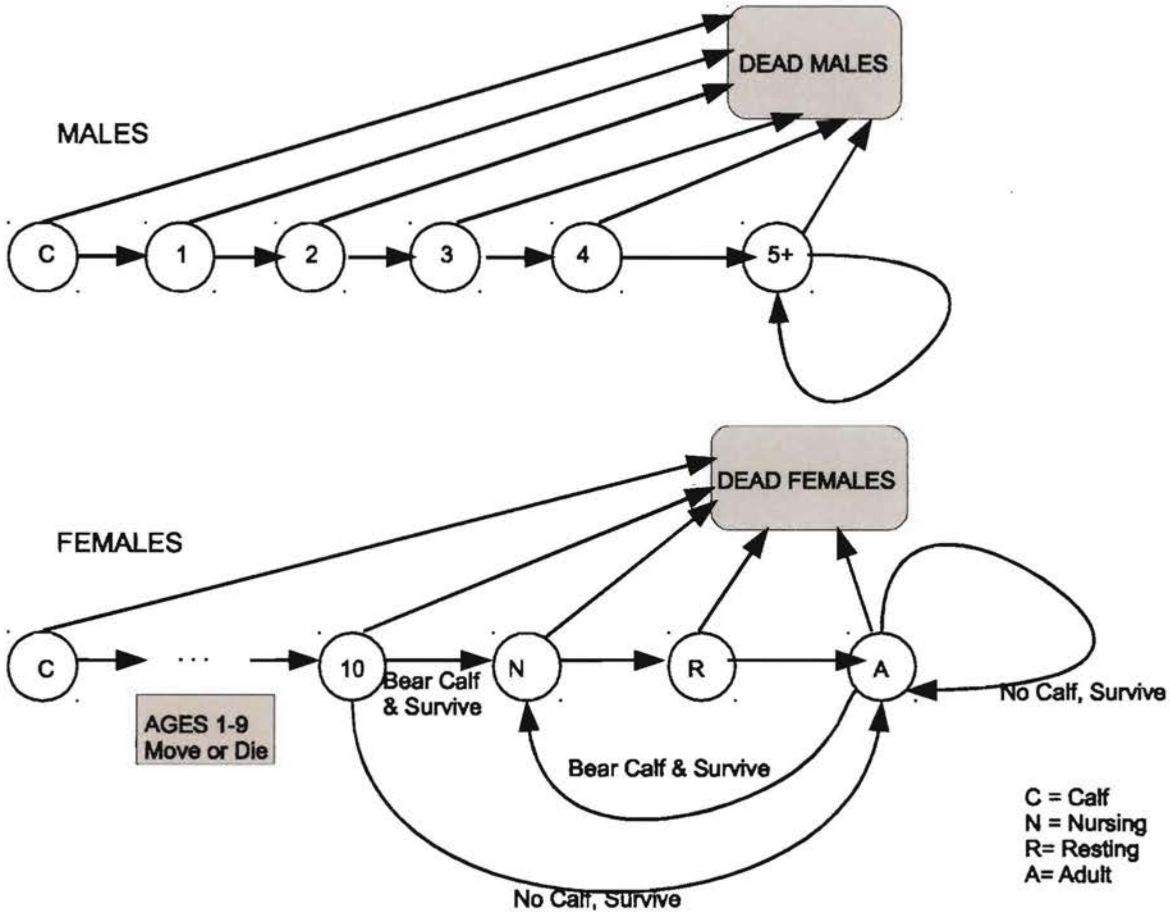
Simple Mortality Reduction ([**B**]1 animal of any class or [**C**] one adult female), and Per Capita Mortality Reductions (rate equivalent to [**D**] 1/345 any class, [**E**] 1/adult female class, [**F**] 3/345 any class). Overall growth, lambda, was calculated as Lambda = $e^{((\log_e(N.\text{final})-\log_e(345))/100)}$, where N.final was the ending population size.

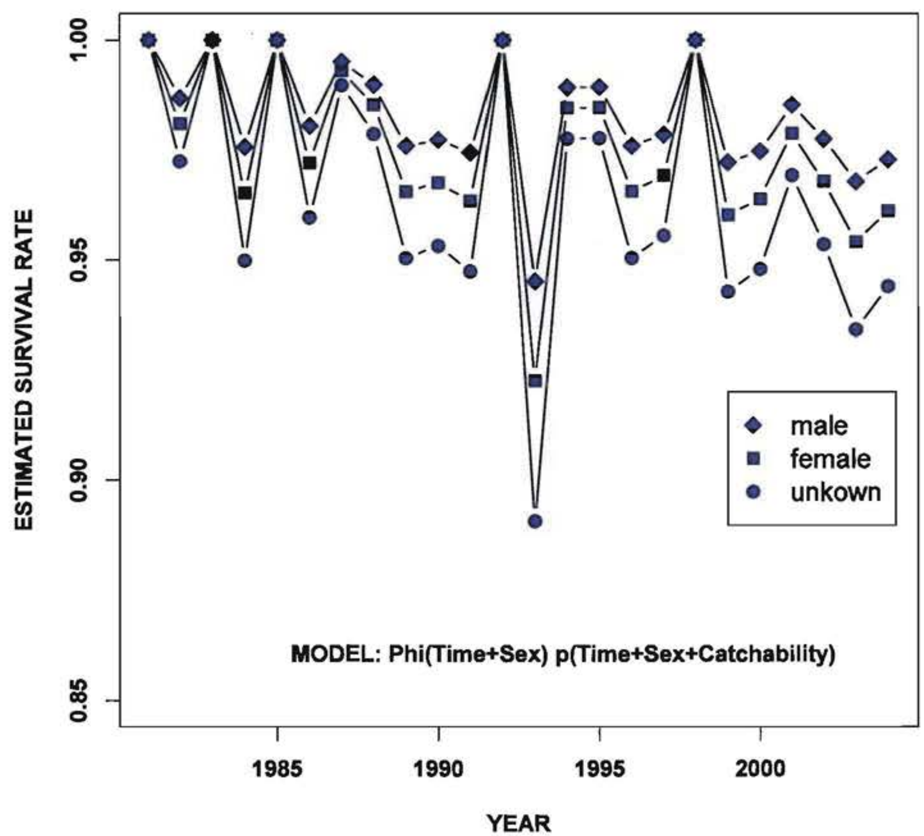
**Fig. 6**. Empirical cumulative distributions of lambda (100 years) and lambda.35 (35 Years) from 1000 replicated population dynamic scenarios for North Atlantic right whales for which the amount of human caused mortality was reduced: Status quo (no reduction), Simple Mortality Reduction (1 animal of any class or one adult female), and Per Capita Mortality Reductions (rate equivalent to 1/345 any class, 1/adult female class, 3/345 any class). Graphs depict the cumulative proportion of projections that exceed the 2% growth rate threshold necessary to declare recovery.

Figure 1



MALES
DEAD MALES
C
1
2
3
4
5+
FEMALES
DEAD FEMALES
C
...
10
N
R
A
Bear Calf & Survive
AGES 1-9 Move or Die
Bear Calf & Survive
No Calf, Survive
No Calf, Survive
C = Calf
N = Nursing
R= Resting
A= Adult

Figure 2



1.00
0.95
0.90
0.85
ESTIMATED SURVIVAL RATE
1985
1990
1995
2000
YEAR
male
female
unkown
MODEL: Phi(Time+Sex) p(Time+Sex+Catchability)

Figure 3

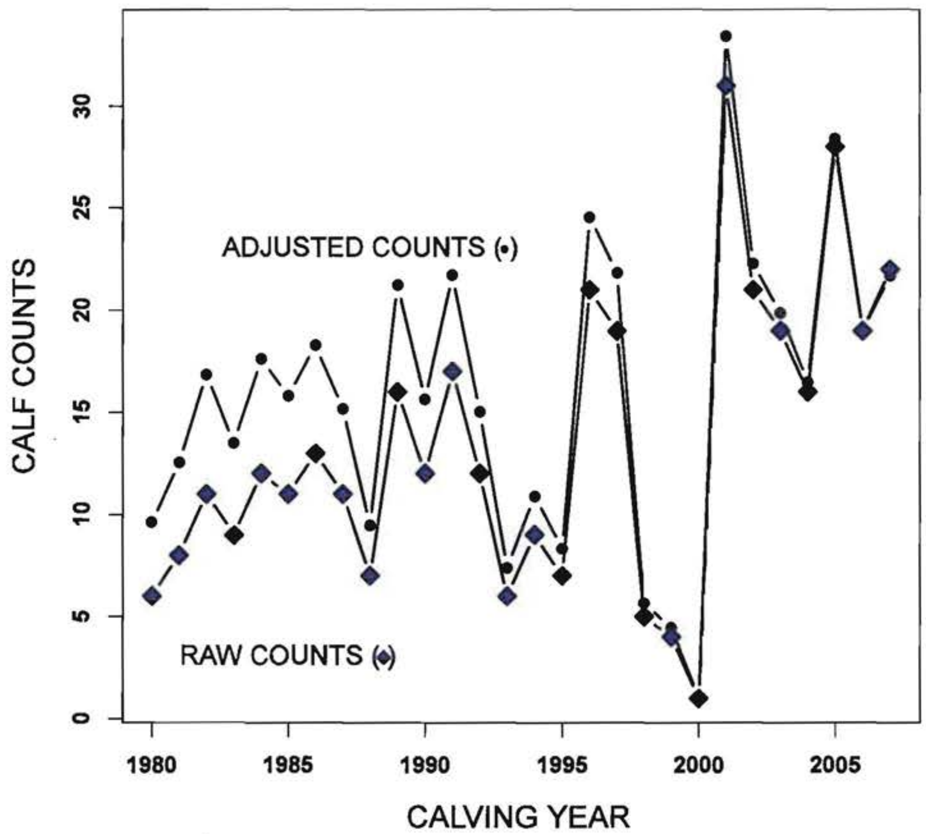

ADJUSTED COUNTS (•)
RAW COUNTS (◆)
CALF COUNTS
0
5
10
15
20
25
30
1980
1985
1990
1995
2000
2005
CALVING YEAR

Figure 4

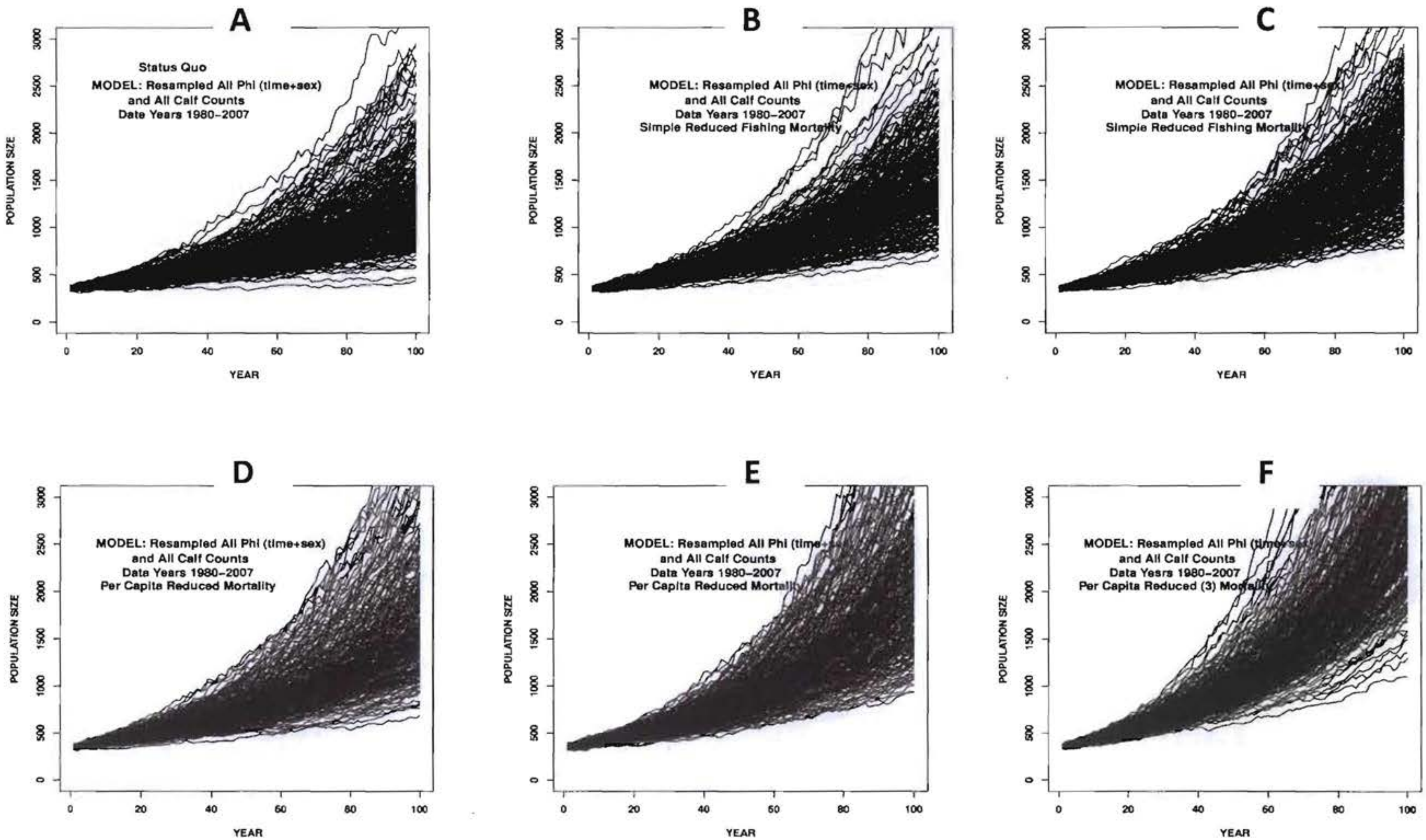
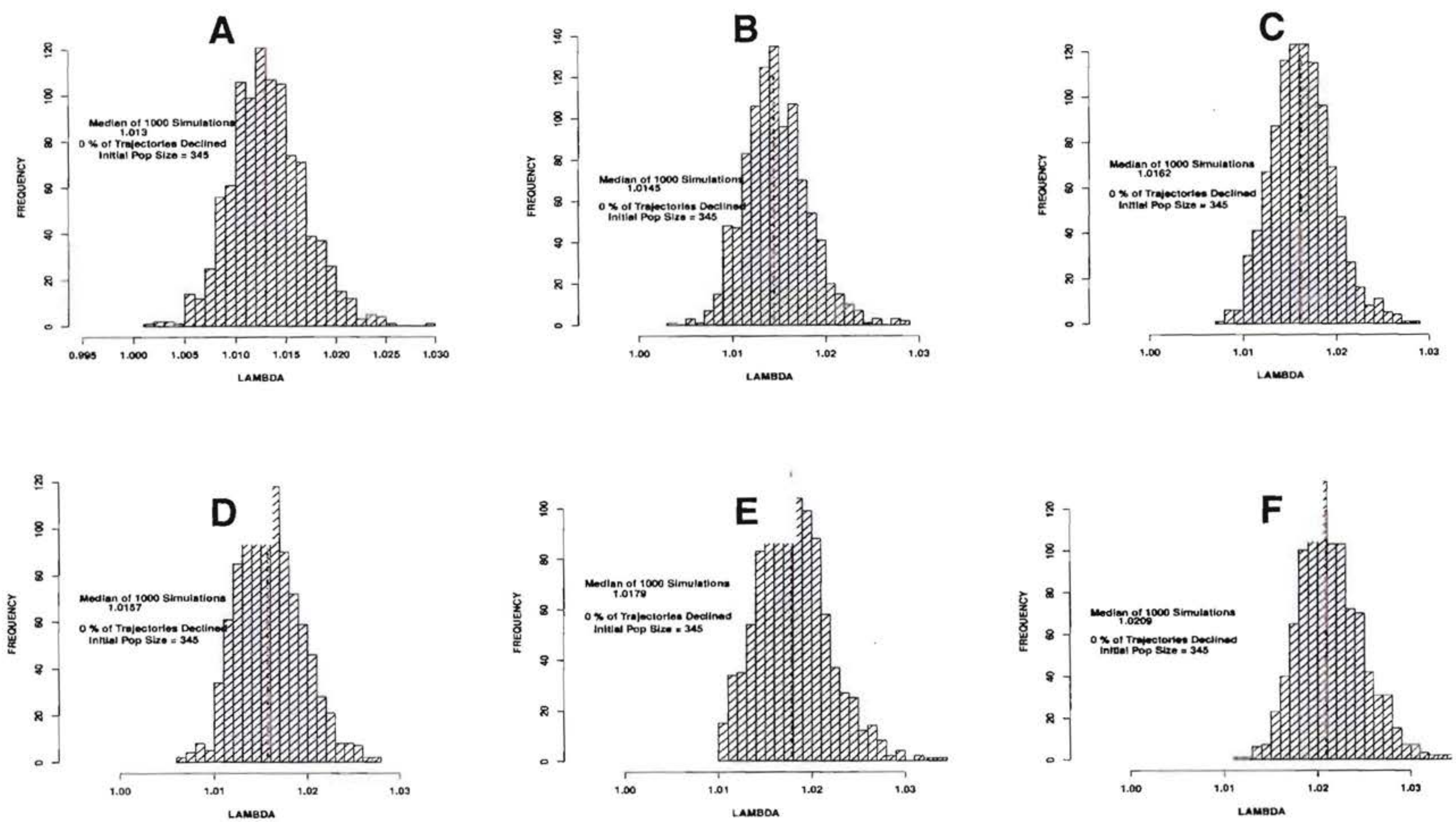
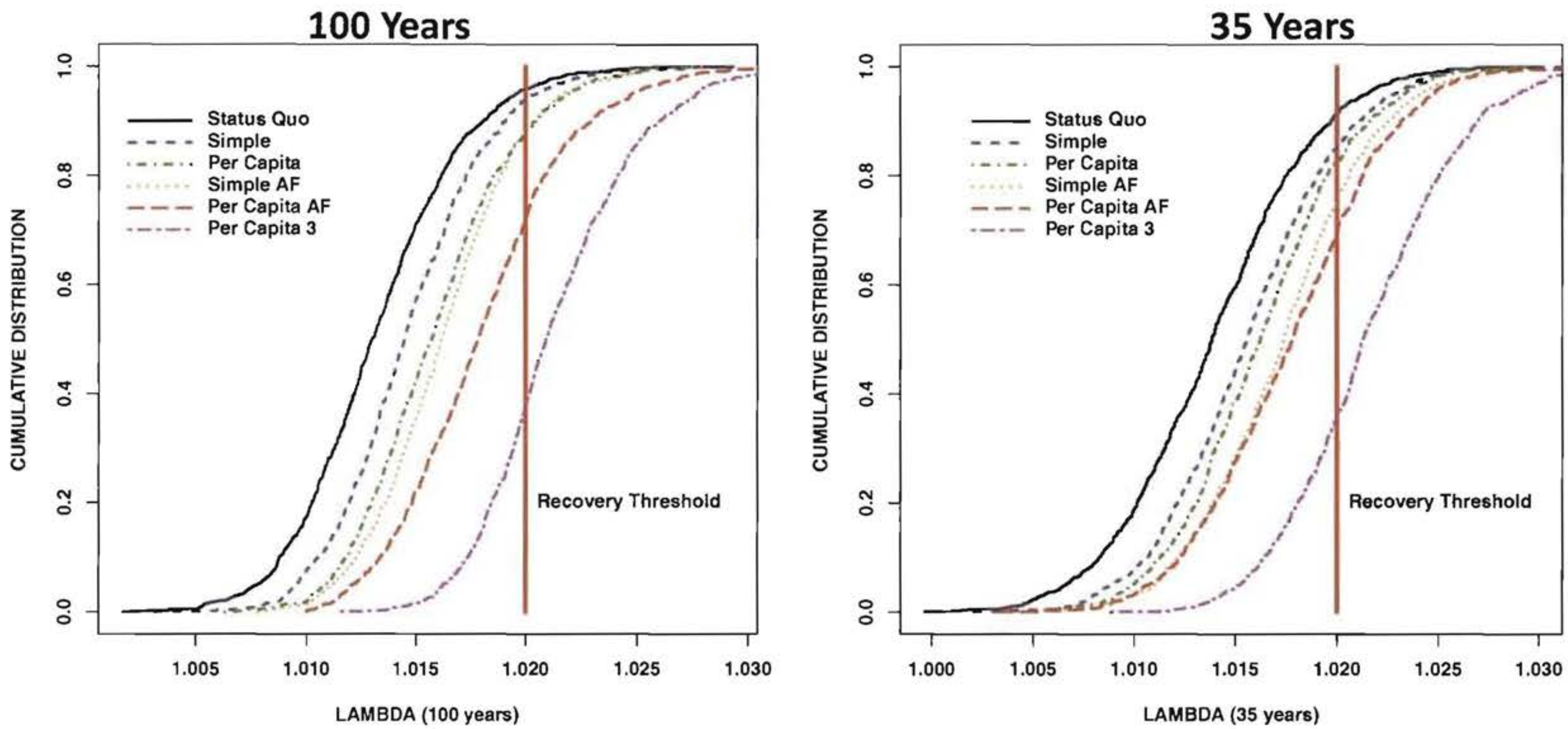

A
Status Quo
MODEL: Resampled All Phi (time+sex)
and All Calf Counts
Date Years 1980-2007
B
MODEL: Resampled All Phi (time+sex)
and All Calf Counts
Data Years 1980-2007
Simple Reduced Fishing Mortality
C
MODEL: Resampled All Phi (time+sex)
and All Calf Counts
Data Years 1980-2007
Simple Reduced Fishing Mortality
D
MODEL: Resampled All Phi (time+sex)
and All Calf Counts
Data Years 1980-2007
Per Capita Reduced Mortality
E
MODEL: Resampled All Phi (time+sex)
and All Calf Counts
Data Years 1980-2007
Per Capita Reduced Mortality
F
MODEL: Resampled All Phi (time+sex)
and All Calf Counts
Data Years 1980-2007
Per Capita Reduced (3) Mortality
POPULATION SIZE
YEAR

Figure 5



A
Median of 1000 Simulations
1.013
0 % of Trajectories Declined
Initial Pop Size = 345
FREQUENCY
LAMBDA
B
Median of 1000 Simulations
1.0145
0 % of Trajectories Declined
Initial Pop Size = 345
FREQUENCY
LAMBDA
C
Median of 1000 Simulations
1.0162
0 % of Trajectories Declined
Initial Pop Size = 345
FREQUENCY
LAMBDA
D
Median of 1000 Simulations
1.0157
0 % of Trajectories Declined
Initial Pop Size = 345
FREQUENCY
LAMBDA
E
Median of 1000 Simulations
1.0179
0 % of Trajectories Declined
Initial Pop Size = 345
FREQUENCY
LAMBDA
F
Median of 1000 Simulations
1.0209
0 % of Trajectories Declined
Initial Pop Size = 345
FREQUENCY
LAMBDA

Figure 6



100 Years
Status Quo
Simple
Per Capita
Simple AF
Per Capita AF
Per Capita 3
Recovery Threshold
CUMULATIVE DISTRIBUTION
0.0
0.2
0.4
0.6
0.8
1.0
1.005
1.010
1.015
1.020
1.025
1.030
LAMBDA (100 years)
35 Years
Status Quo
Simple
Per Capita
Simple AF
Per Capita AF
Per Capita 3
Recovery Threshold
CUMULATIVE DISTRIBUTION
0.0
0.2
0.4
0.6
0.8
1.0
1.000
1.005
1.010
1.015
1.020
1.025
1.030
LAMBDA (35 years)

# 17.0 APPENDIX B: Analysis of Atlantic Sea Scallop Fishery Impacts on the North Atlantic Population of Loggerhead Sea Turtles

NOAA Technical Memorandum NMFS-NE-207

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

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

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**NOAA Technical Memorandum NMFS-NE-207**

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

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**National Oceanic and Atmospheric Administration**
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**National Marine Fisheries Service**
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**Northeast Fisheries Science Center**
**Woods Hole**

**February 2008**

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# ABSTRACT

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

# INTRODUCTION

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

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

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

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

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

---

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

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

# METHODS

## Data

### *Population trend data*

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

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

### *Current abundance data*

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

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

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

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

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

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

## *Fishery mortality data*

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

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

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

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

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

where:

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

$F_M$ = proportion of bycatch mature

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

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

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

$F_L$ = proportion of the bycatch considered as lethal takes

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

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

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

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

## Model

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

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

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

$\mu$ – the arithmetic mean of the log population growth rate
$\sigma^2$ – variance of the log population growth rate

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

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

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

Then $\mu$ was calculated as:

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

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

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

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

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

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

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

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

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

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

## Modeling Steps

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

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

## Evaluation of Results

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

# RESULTS

## Population Trends to Present

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

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

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

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

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

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

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

## Viability Analyses

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

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

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

## Model Sensitivity

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

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

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

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

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

## DISCUSSION

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

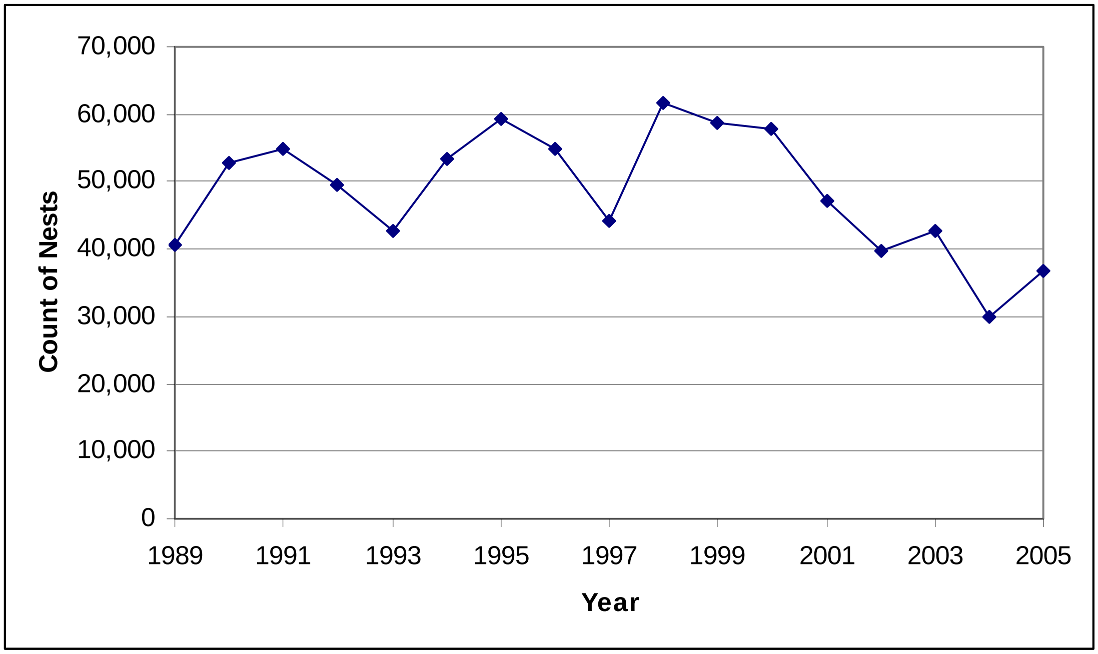


Figure 1. Number of Atlantic Ocean loggerhead sea turtle (*Caretta caretta*) nests recorded at US Northern (NC, SC, GA) and Peninsular Florida index beaches from 1989 to 2005 (NMFS in review, FWRI 2007).

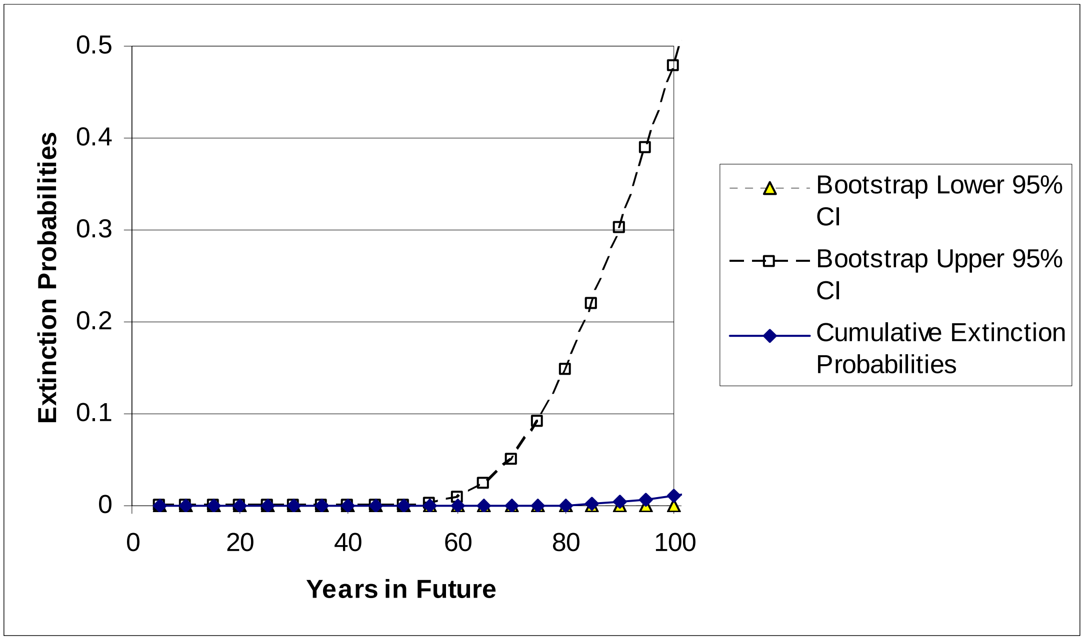


Figure 2. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 base model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)*. Quasi-extinction is equal to 250 adult female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

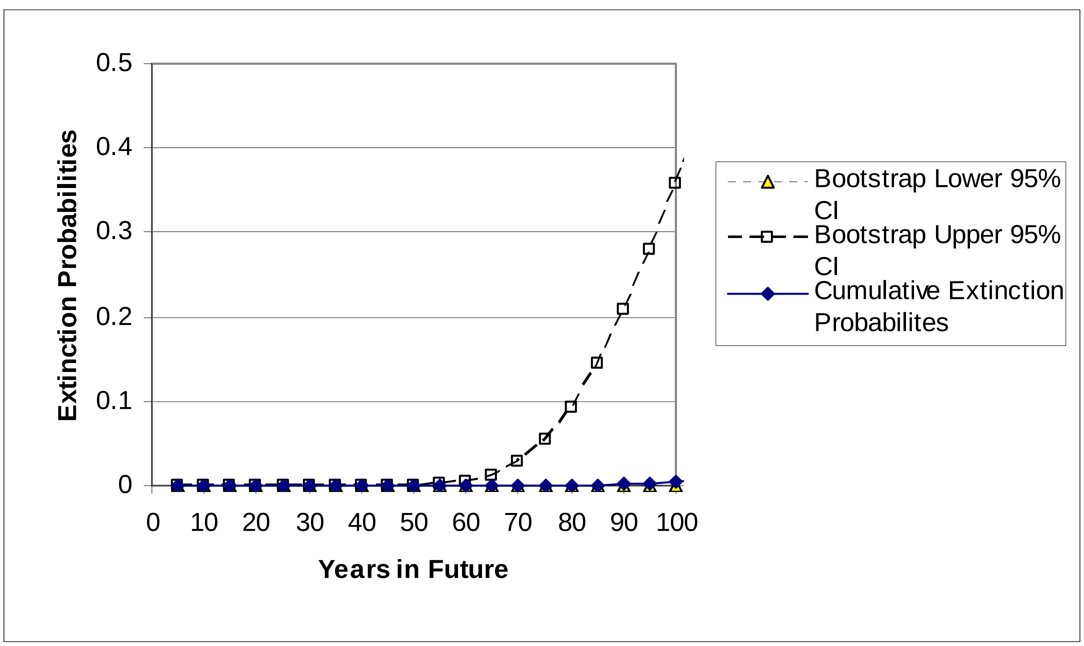


Figure 3. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)* added back into population. Quasi-extinction is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

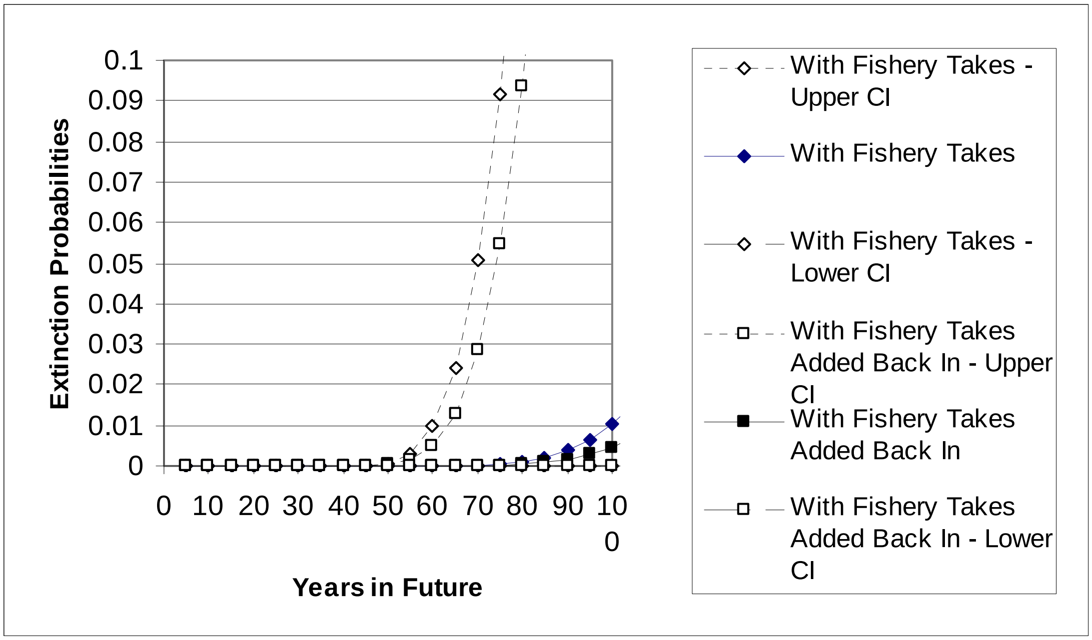


Figure 4. Comparison of cumulative quasi-extinction probabilities and confidence intervals (CI) of 1989-2005 models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta)*. Note vertical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

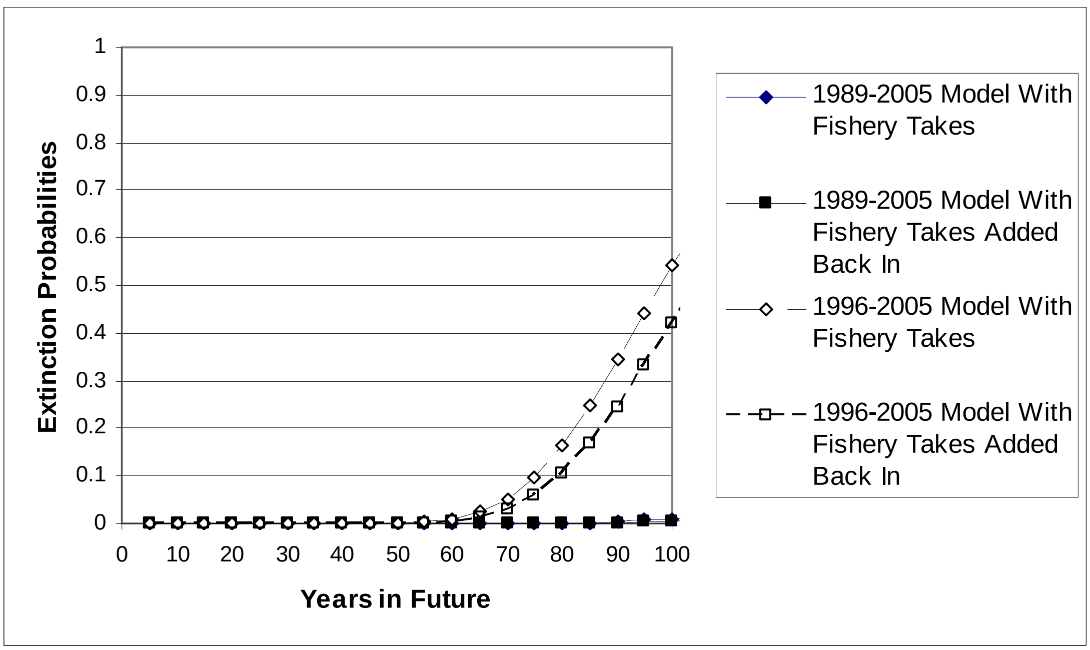


Figure 5. Extinction trajectories for models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes with original 1989-2005 population trajectory compared to 1996-2005 trajectory. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta)*.

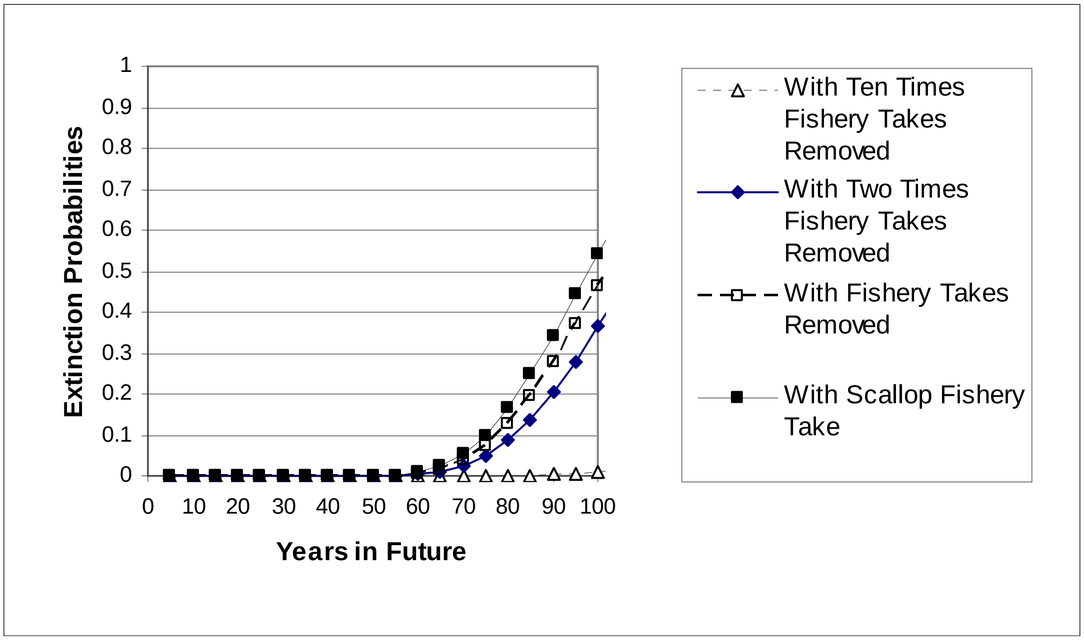


Figure 6. Cumulative quasi-extinction probabilities for 1996-2005 models with various levels of mortality removed from the trend. Fishery takes estimated as one time (the Atlantic sea scallop [*Placopecten magellanicus*] fisheries) versus two and ten times the original sea scallop fishery take level.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

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# 18.0APPENDIX C: Sea Turtle Resuscitation Measures

**Sea turtle and resuscitation measures as found at 50 CFR 223.206(d)(1).**
(d) (1) (i) Any specimen taken incidentally during the course of fishing or scientific research activities must be handled with due care to prevent injury to live specimens, observed for activity, and returned to the water according to the following procedures.

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

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

(1) placing the turtle on its bottom shell (plastron) so that the turtle is right side up, and elevating its hindquarters at least 6 inches (15.2 cm) for a period of 4 up to 24 hours. The amount of the elevation depends on the size of the turtle; greater elevations are needed for larger turtles. Periodically, rock the turtle gently left to right and right to left by holding the outer edge of the shell (carapace) and lifting one side about 3 inches (7.6 cm) then alternate to the other side. Gently touch the eye and pinch the tail (reflex test) periodically to see if there is a response.
(2) sea turtles being resuscitated must be shaded and kept damp or moist but under no circumstance be placed into a container holding water. A water-soaked towel placed over the head, neck, and flippers is the most effective method in keeping a turtle moist.
(3) sea turtles that revive and become active must be released over the stern of the boat only when fishing or scientific collection gear is not in use, when the engine gears are in neutral position, and in areas where they are unlikely to be recaptured or injured by vessels. Sea turtles that fail to respond to the reflex test or fail to move within 4 hours (up to 24, if possible) must be returned to the water in the same manner as that for actively moving turtles.

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

## 19.0 APPENDIX D: Procedure for Obtaining Fin Clips from Atlantic Sturgeon for Genetic Analysis

*Obtaining Sample*

1. Wash hands and use disposable gloves. Ensure that any knife, scalpel, or scissors used for sampling has been thoroughly cleaned and wiped with alcohol to minimize the risk of contamination.

2. For any sturgeon, after the specimen has been measured and photographed, take a one-cm square clip from the pelvic fin.

3. Each fin clip should be placed into a vial of 95% non-denatured ethanol and the vial should be labeled with the species name, date, name of project and the fork length and total length of the fish along with a note identifying the fish to the appropriate observer report. All vials should be sealed with a lid and further secured with tape Please use permanent marker and cover any markings with tape to minimize the chance of smearing or erasure.

*Storage of Sample*

1. If possible, place the vial on ice for the first 24 hours. If ice is not available, please refrigerate the vial. Send as soon as possible as instructed below.

*Sending of Sample*

1. Vials should be placed into Ziploc or similar resealable plastic bags. Vials should be then wrapped in bubble wrap or newspaper (to prevent breakage) and sent to:

Julie Carter
NOAA/NOS – Marine Forensics
219 Fort Johnson Road
Charleston, SC 29412-9110
Phone: 843-762-8547

Prior to sending the sample, contact Lynn Lankshear at NMFS Northeast Regional Office (978-282-8473) to report that a sample is being sent and to discuss proper shipping procedures.

## 20.0 APPENDIX E: Procedure for Obtaining Fin Clips from Atlantic Salmon for Genetic Analysis

This procedure has been amended from the "GENETIC SAMPLING PROCEDURE (Standard Operating Procedure R-07)" instructions documented by the Population Dynamics Branch of the Northeast Fisheries Science Center of NOAA Fisheries.

**Equipment needed**:

1. Cooler and cold ice packs or wet ice.
2. Pre labeled vials
3. Ethyl alcohol (ethanol)
4. Fin clippers, dermal punches and probe (i.e. section of wire, paper clip, cake tester, etc.) or scissors
5. Forceps

**Sampling:**

1. Flush the area to be clipped with sea water and rinse with distilled water if available.
   Carefully clip or dermal punch a small **(3mm x 3mm)** section of the anal, upper or lower caudal fin (depending on clipping schedule – see **Temporary Marking Procedures (Fin Clip and Punch; SOPs R-05 and R-06)**When clipping the fin remember to include rays along with the cartilage.
2. Place the section of fin into a labeled vial containing ethanol, and cap it. The amount of alcohol to use per sample should be at least 3:1 liquid/tissue ratio; less would greatly diminish the alcohol's ability to preserve the tissue.
3. Make sure you indicate the vial # on the datasheet.
4. Place sample on ice and out of sun and rain.
5. Clean the fin clippers/dermal punch between samples in sea water or distilled water.
6. Transfer sample vials to refrigerator when back at office/field station.
   i. Label individual vials with internal and external labels which contain a JoinID. Be sure to secure the label with a small piece of tape connecting the ends of the label so that the label stays on the vial.

**Things to think about:**

1. Minimize stress on the fish by holding it gently but in a manner such that it cannot move. This is best done by holding as much of the fish in your hands as possible (i.e., do not hold only the front or only the back of the fish – place your hands around the entire fish).
2. Minimize stress on the fish by performing this procedure as quickly as possible. It is important to ascertain how the clipper wants the fish

presented (held) to them *before* the fish is taken from the water, and preferably, before the fish is taken from the holding area of the trap.

**Storage and Sending of Sample:**

1. If possible, place the vial on ice for the first 24 hours. If ice is not available, please refrigerate the vial. Send as soon as possible as instructed below.

2. Vials should be placed into Ziploc or similar resealable plastic bags. Vials should be then wrapped in bubble wrap or newspaper (to prevent breakage) and sent to:

Julie Carter
NOAA/NOS – Marine Forensics
219 Fort Johnson Road
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