

Endangered Species Act – Section 7 Consultation


Biological Opinion

Action Agency: National Oceanic and Atmospheric Administration (NOAA),
National Marine Fisheries Service (NMFS), Highly Migratory
Species (HMS) Division

Activity: Endangered Species Act (ESA) Section 7 Consultation on the
Operation of the HMS Fisheries (Excluding Pelagic Longline)
under the Consolidated Atlantic HMS Fishery Management Plan
(F/SER/2015/16974)

Consulting Agency: NOAA, NMFS, Southeast Regional Office (SERO), Protected
Resources Division

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List of Frequently Used Acronyms

ACCSP	Atlantic Coastal Cooperative Statistics Program
ALWTRP	Atlantic Large Whale Take Reduction Plan
ALWTRT	Atlantic Large Whale Take Reduction Team
ASMFC	Atlantic States Marine Fisheries Commission
CCL	Curved Carapace Length
COE	United States Army Corps of Engineers
CPUE	Catch Per Unit Effort
CSFOP	Commercial Shark Fishery Observer Program
DAM	Dynamic Area Management
DPS	Distinct Population Segment
DWH	Deepwater Horizon
EFP	Exempted Fishing Permit
EEZ	Exclusive Economic Zone
EPA	Environmental Protection Agency
ESA	Endangered Species Act of 1973
FMP	Fishery Management Plan
HMS	Highly Migratory Species
ITS	Incidental Take Statement
LCS	Large Coastal Sharks
LOF	List of Fisheries
LPS	Large Pelagics Survey
MMPA	Marine Mammal Protection Act of 1972
MRIP	Marine Recreational Information Program
MSA	Mixed Stock Analysis
MSFCMA	Magnuson-Stevens Fishery Conservation and Management Act
NEFOP	Northeast Fisheries Observer Program
NER	Northeast Regional Office
NSED	National Sawfish Encounter Database
PCB	Polychlorinatedbiphenyl
PRD	Protected Resources Division
RPA	Reasonable and Prudent Alternatives
SAM	Seasonal Area Management
SAR	Stock Assessment Report
SCL	Straight Carapace Length
SCS	Small Coastal Sharks
SEFSC	Southeast Fisheries Science Center
SERO	Southeast Regional Office
SGOP	Shark Gillnet Observer Program
SI/M	Serious Injury or Mortality
STSSN	Sea Turtle Stranding and Salvage Network
TED	Turtle Excluder Device
TEWG	Turtle Expert Working Group
USCG	United States Coast Guard
USFWS	United States Fish and Wildlife Service
USN	United States Navy
VTR	Vessel Trip Reporting Program
YOY	Young of the Year

Introduction

Section 7(a)(2) of the ESA of 1973, as amended (16 U.S.C. § 1531 et seq.), requires each federal agency to ensure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of any critical habitat of such species. To fulfill this obligation, Section 7(a)(2) requires federal agencies to consult with the appropriate Secretary on any action they propose that “may affect” listed species or designated critical habitat. NMFS and the United States Fish and Wildlife Service (USFWS) share responsibilities for administering the ESA.

A federal action agency requests consultation when it determines that a proposed action “may affect” listed species or designated critical habitat. Consultations on most listed marine species and their designated critical habitat are conducted between the action agency and NMFS and conclude after NMFS concurs with an action agency that its action is not likely to adversely affect listed species or critical habitat, or issues a Biological Opinion (“Opinion”) that identifies whether a proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify its critical habitat. If jeopardy or destruction or adverse modification is found to be likely, the Opinion identifies reasonable and prudent alternatives (RPAs) to the action as proposed, if any, that can avoid jeopardizing listed species or resulting in the destruction/adverse modification of critical habitat. The Opinion states the amount or extent of incidental take of the listed species that may occur, specifies reasonable and prudent measures (RPMs) that are required to minimize the impacts of incidental take and and terms and conditions for implementing those measures, reporting and monitor, and recommends conservation measures to further conserve the species.

As provided in 50 CFR 402.16, reinitiation of formal consultation is required when discretionary involvement or control over the action has been retained (or is authorized by law) and: (1) the amount or extent of the incidental take is exceeded; (2) new information reveals effects of the agency 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 previously considered; or (4) if a new species is listed or critical habitat designated that may be affected by the identified action. NMFS and action agencies have discretion to reinitiate formal consultation in other circumstances as appropriate.

The proposed action encompasses the operation of Atlantic HMS fisheries (excluding the pelagic longline fishery)¹ as carried out under the 2006 Consolidated Atlantic HMS Fishery Management Plan (2006 Consolidated HMS FMP), as amended. This document represents NMFS’ Opinion on the effects of that proposed action on threatened and endangered species and their designated critical habitat, in accordance with Section 7 of the ESA. NMFS has dual responsibilities as both the action agency that authorized the fisheries under the authority of the Magnuson-Stevens Fishery Conservation and Management Act (16 U.S.C. §1801 et seq.) (MSA) and the consulting agency under the authority of the ESA. For the purposes of this consultation, the HMS

¹ The HMS Management Division requested reinitiation of consultation with SERO PRD on the pelagic longline fishery, also managed under the 2006 Consolidated HMS FMP, on March 31, 2014.

Management Division is considered the action agency and the consulting agency is the Southeast Regional Office (SERO) Protected Resources Division (PRD).

We, SERO PRD, have prepared this opinion in accordance with Section 7 of the ESA and regulations promulgated to implement that section of the ESA. It is based on information provided in the 2006 Consolidated HMS FMP and subsequent amendments to the HMS FMP, biological evaluations from the HMS Management Division, status reviews, recovery plans, research, population modeling efforts, and other relevant published and unpublished scientific and commercial data cited in the Literature Cited section of this document.

1.0 Consultation History

Since the 1980s, fisheries targeting Atlantic HMS have undergone many formal and informal ESA section 7 consultations to evaluate their effects on threatened and endangered species and to ensure that actions proceed in a way that complies with the requirements of the ESA. Prior to approval and implementation of the 2006 Consolidated HMS FMP, NMFS consulted on fisheries targeting Atlantic HMS as managed under the 1999 Atlantic HMS FMP (1999 HMS FMP) and the 1999 Atlantic Billfish FMP (Billfish FMP), under authority of MSA. Prior to that, Atlantic swordfish, shark and billfish were all consulted on as carried out under the separate FMPs,² under authority of the MSA, and Atlantic tunas were managed only under authority of the Atlantic Tunas Convention Act (ATCA). Consultations on HMS-authorized fisheries targeting tunas, swordfish, sharks, and billfish prior to 2001 are summarized in a June 30, 2000 Opinion and a June 14, 2001 Opinion.

The last comprehensive Section 7 evaluation of the effects of all Atlantic HMS fisheries on ESA-listed species was the June 14, 2001 Opinion. Since completing that consultation, NMFS has undertaken additional formal consultation on certain HMS fisheries. Consequently, most of the 2001 Opinion has been superseded by other consultations. Below we summarize the 2001 Opinion and opinions completed since then, to provide context for the current consultation. Each of the opinions discussed below, i.e., the 2001, 2003, 2004, 2008, and the 2012 Opinions) include more detailed consultation histories for each consultation.

The 2001 Opinion on the Reinitiation of Consultation on the Atlantic HMS FMP and its Associated Fisheries (hereafter, the 2001 Opinion) analyzed the impacts of the pelagic longline fishery, the Southeast U.S. shark drift gillnet fishery, the bottom longline fishery for sharks, and the additional HMS fisheries (*i.e.*, tuna purse seine, harpoon/hand gear fisheries, hook-and-line, *etc.*). In addition to considering new information on sea turtle interactions and sea turtle status, the consultation considered the effects of several regulatory changes: the implementation of the bycatch reduction regulatory amendment with an August 1, 2000, final rule; the October 13, 2000, emergency rule on the pelagic longline fishery that temporarily closed an area off the Grand Banks; and the interim final rule requiring pelagic longline vessels to carry and use line clippers and dipnets.

The 2001 Opinion concluded that the operation of the pelagic longline fishery was likely to jeopardize the continued existence of loggerhead and leatherback sea turtles. All other HMS fisheries, including the Atlantic shark bottom longline and gillnet fisheries, were found to adversely affect but not be likely to jeopardize the continued existence of any ESA-listed species. The 2001 Opinion specified a reasonable and prudent alternative (RPA) which would

² In 1985 and 1988, the five Atlantic-based Fishery Management Councils finalized an Atlantic Swordfish FMP and Atlantic Billfish FMP, respectively. In 1993, NMFS implemented the FMP for Sharks of the Atlantic Ocean (1993 Shark FMP). In 1999, NMFS combined the 1993 Shark FMP and the Atlantic Swordfish FMP into a single FMP, the 1999 HMS FMP. This new FMP also encompassed existing Atlantic tunas regulations. Atlantic billfish continued to be managed under a separate FMP. In 2006, NMFS consolidated the management of Atlantic billfish with that of swordfish, tunas, and sharks into one comprehensive FMP, the 2006 Consolidated HMS FMP.

allow the operation of the pelagic longline fishery without jeopardizing the continued existence of loggerhead and leatherback sea turtles.

On July 9, 2002, NOAA Fisheries published the final rule (67 FR 45393) implementing all of the measures identified in the 2001 Opinion RPA to reduce the incidental catch and post-release mortality of sea turtles and other protected species in HMS pelagic longline, bottom longline, and gillnet fisheries. The rule implemented the closure of the Northeast Distant statistical reporting area, required the length of any gangion to be 10 percent longer than the length of any floatline if the total length of any gangion plus the total length of any floatline was less than 100 meters, prohibited vessels from having hooks on board other than corrodible, nonstainless steel hooks, and required all HMS bottom and pelagic longline vessels to post sea turtle handling and release guidelines in the wheelhouse. The final rule additionally established regulations for the HMS shark gillnet fishery that required additional measures as follows: both the observer and vessel operator to look for whales; the vessel operator to contact NMFS if a listed whale was taken; and shark gillnet fishermen to conduct net checks every 0.5 to 2 hours to look for and remove any sea turtles or marine mammals from their gear. NMFS did not implement the gangion placement requirement because it was found to result in an unchanged number of interactions with loggerhead sea turtles and an apparent increase in interactions with leatherback sea turtles.

On August 1, 2003, NMFS published a proposed rule for Draft Amendment 1 to the 1999 HMS FMP. Amendment 1 dealt exclusively with measures affecting the management of sharks and the directed shark fishery components (i.e., bottom longline, Southeast shark drift gillnet, and recreational shark fisheries) of the 1999 HMS FMP. NMFS consulted on the effects of the directed shark fisheries on listed species based on new information obtained subsequent to the 2001 Opinion, as well as to address potential adverse effects from shark fisheries on the newly listed smalltooth sawfish (*Pristis pectinata*). The proposed rule and new information, as well as the effects on smalltooth sawfish were limited to directed shark fisheries and did not affect pelagic longline fishing effort or other fishing patterns previously analyzed in the 2001 Opinion. Therefore, the scope of the consultation was limited to the directed shark fisheries.

On October 29, 2003, SERO PRD completed its new Opinion on the operation of Atlantic shark fisheries under the 1999 HMS FMP and Amendment 1. The 2003 Opinion concluded that the operation of the Atlantic shark fisheries was not likely to jeopardize the continued existence, or destroy or adversely modify critical habitat, of any ESA-listed species. A 5-year ITS was included that specified the extent of anticipated take of sea turtles and smalltooth sawfish and the RPMs necessary to minimize the impacts of the anticipated take: 172 leatherback sea turtles of which 88 would be lethal; 1370 loggerhead sea turtles of which 755 would be lethal; 30 total in any combination of hawksbill, green, and Kemp's ridley sea turtles (with 5 lethal takes per species); and 261 smalltooth sawfish, of which no lethal takes were expected. For the directed Atlantic shark fisheries only, the 2003 Opinion superseded the 2001 Opinion.

On June 1, 2004, NMFS completed an Opinion evaluating the effects on listed species by the Atlantic HMS pelagic longline fishery: (1) as it was currently being prosecuted, including fishing under exempted fishing permits (EFPs) and scientific research permits (SRPs); and (2) as it would be prosecuted under the proposed regulations that required new sea turtle bycatch and mortality reduction measures (i.e., hook and bait requirements, gear removal and handling

requirements) (NMFS 2004). The effects of the proposed rule to implement the 2002 International Commission for the Conservation of Atlantic Tunas (ICCAT) swordfish quota recommendations were also evaluated in this consultation. The proposed regulatory actions were specific to the HMS pelagic longline fishery and not any of the other fisheries under the 1999 HMS FMP or Billfish FMP. There was no new information suggesting the manner or extent of effects to any listed species from the remaining fisheries under the 1999 HMS FMP (i.e. purse seine, harpoon, hand line, rod-and-reel fisheries) had changed. Consequently, consultation was limited to the HMS pelagic longline fishery and the scope listed above.

The 2004 Opinion found that the operation of the Atlantic HMS pelagic longline fishery as proposed was likely to jeopardize the continued existence of leatherback sea turtles; however, the Opinion stated that the Atlantic HMS pelagic longline fishery was not likely to jeopardize the continued existence of loggerhead, green, hawksbill, Kemp's ridley, or olive ridley sea turtles. The Opinion established an RPA in order to avoid jeopardizing leatherback sea turtles, which included, among other things, maximization of gear removal, a comprehensive outreach program to ensure that fishermen were made aware of the safe handling and gear removal requirements, and a net mortality rate performance standard and requirements to improve monitoring requirements to verify maximized gear removal and predict anticipated total mortality. The Opinion stated that the RPA would also benefit loggerhead sea turtles and that, where those benefits affected the anticipated impact on loggerhead sea turtles in a quantifiable way, those reduced impacts were included in the RPA. Thus, the RPA also provided a net mortality rate performance standard and an estimate of anticipated total mortality level for loggerhead sea turtles. For the Atlantic HMS pelagic longline fishery, the 2004 opinion superseded the 2001 Opinion.

Consultation solely on Atlantic shark fisheries managed under the 2006 Consolidated HMS FMP was conducted formally two more times. On May 20, 2008, NMFS completed formal consultation on the Atlantic shark fisheries and proposed amendments to the commercial and recreational regulations governing shark fisheries in the Atlantic, Gulf of Mexico, and Caribbean Sea (NMFS 2008c). The Opinion concluded that the operation of the shark fisheries (Commercial Shark Bottom Longline, Commercial Shark Gillnet, and Recreational Shark Handgear Fisheries) as managed under the 2006 Consolidated HMS FMP, including Amendment 2, was not likely to jeopardize the continued existence of green, hawksbill, Kemp's ridley, leatherback, or loggerhead sea turtles or smalltooth sawfish. An ITS was issued specifying the amount and extent of anticipated take on a three-year basis, along with RPMs and associated terms and conditions deemed necessary and appropriate to minimize the impact of these takes. Other listed species were found to be not likely to be adversely affected. No critical habitat overlapped with the action area, thus none was affected.

On May 20, 2012, NMFS completed the most recent formal consultation on the Atlantic shark fisheries (NMFS 2012c). The consultation addressed potential effects of federal management for smoothhound shark. The Opinion evaluated the effects of the shark fisheries carried out under the 2006 Consolidated HMS FMP including the existing components of the Atlantic shark fisheries (i.e., bottom longlines and gillnets), as well as the new smoothhound fishery (i.e., a gillnet fishery), on ESA listed species, including a new listed species, Atlantic sturgeon, which was adversely affected only by the new smoothhound gillnet component. The Opinion

(hereafter, the 2012 Opinion) concluded that the proposed action was not likely to jeopardize the continued existence of North Atlantic right, humpback, and fin whales, green, hawksbill, Kemp's ridley, leatherback, or loggerhead sea turtles, smalltooth sawfish, or Atlantic sturgeon. NMFS anticipated take and included a three-year ITS of 126 loggerhead sea turtles, 57 green sea turtles, 18 leatherback sea turtles, 36 Kemp's ridley sea turtles, 18 hawksbills, 32 smalltooth sawfish, 321 Atlantic Sturgeon from five Distinct Population Segments (DPSs). Other listed species and critical habitat were found to be not likely to be adversely affected.

On March 31, 2014, the HMS Management Division requested reinitiation of Section 7 consultation on the operation of the Atlantic HMS pelagic longline fishery. Reinitiation was requested based on the availability of new information revealing effects of the action that may affect listed species in a manner or to an extent not previously considered (see 50 C.F.R. § 402.16(b)). Specifically, the request was based on information indicating that the net mortality rate and total mortality estimates for leatherback sea turtles specified in the 2004 Opinion's reasonable and prudent alternative were exceeded (although the take level specified in the incidental take statement has not been exceeded), changes in information about leatherback and loggerhead sea turtle populations, and new information about sea turtle mortality associated with pelagic longline gear. That consultation is on-going.

This Consultation

On October 30, 2014, the HMS Management Division requested reinitiation of consultation on the operation of Atlantic HMS fisheries as carried out under the 2006 Consolidated HMS FMP (as amended to date) that had previously consulted on in the 2001, 2003, 2008, and the 2012 Opinions (i.e., on all on-going fisheries/gear operations managed under the 2006 Consolidated HMS FMP) except for the pelagic longline fishery, which was separately consulted on in 2004 and was already undergoing separate consultation at that time (see above). The HMS Management Division requested reinitiation of consultation to address potential effects on certain newly listed species, namely the Central and Southwest Atlantic distinct population segment of scalloped hammerhead shark and seven species of corals. NMFS had published, on July 3, 2014, the Final Rule to list the Central and Southwest Atlantic DPS of Scalloped Hammerhead Shark (*Sphyrna lewini*) as Threatened Species (79 FR 38213), and, on August 27, 2014, the Final Rule to list various coral species in the Caribbean, including Florida and the Gulf of Mexico, as threatened (79 FR 53852). The HMS Management Division requested reinitiation because they had determined that the newly listed species identified above occur within the management area of the 2006 HMS Consolidated FMP and may be affected by the operation of these fisheries. Specifically, the HMS Management Division determined that certain authorized Atlantic HMS gear types may affect and are likely to adversely affect scalloped hammerhead sharks within the Central and Southwest Atlantic DPS. Additionally, certain authorized Atlantic HMS gear types may affect but are not likely to adversely affect, threatened Caribbean coral species. These gear types include bandit gear, bottom longline, buoy gear, handline, and rod and reel. The HMS Management submitted a biological evaluation with the request.

On July 8, 2015, the HMS Management Division provided a revised biological evaluation based on further review of the final rule that listed Central and Southwest Atlantic DPS of scalloped hammerhead sharks. The HMS Management Division clarified that use of bottom longline gear and gillnet gear does not occur within the range of that DPS. From 2008-2013, there was no

reported use of these gear types by HMS permit holders in the Caribbean. Several year-round time and area closures in the Caribbean limit use of these gear types. As a result, the HMS Management Division determined that these gear types would have no effect on the Central and Southwest Atlantic DPS of scalloped hammerhead sharks. However, recreational rod and reel was still believed to result in some interactions with these species. That same day, in a memorandum from NMFS Office of Sustainable Fisheries to SERO, the HMS Management Division determined that allowing the operation of all Atlantic HMS fisheries (other than the pelagic longline fishery) during the re-initiation period would not violate Sections 7(a)(2) or 7(d) of the ESA with respect to the Central and Southwest Atlantic DPS of scalloped hammerhead shark and threatened coral species that occur in the action area.

Additional ESA listings and designations took place that affected this consultation. On July 10, 2014, NMFS published a final rule (79 FR 39856) designating critical habitat for the northwest Atlantic Ocean (NWA) loggerhead sea turtle DPS (). Listing actions pertinent to the Atlantic EEZ, other than the coral and scalloped hammerhead shark listings, are as follows. On April 6, 2016, NMFS and the Fish and Wildlife Service (FWS) published a Final Rule (81 FR 20058) removing the range-wide and breeding population ESA listings of the green sea turtle and, in their place, listing 8 green sea turtle DPSs as threatened and 3 green sea turtle DPSs as endangered, effective May 6, 2016. Two of the green sea turtle DPSs, the North Atlantic DPS and the South Atlantic DPS, occur in the South Atlantic Region and were identified as “may be affected” by HMS fishing, based on the earlier 2001, 2003, 2008 and 2012 Opinion analyses for green sea turtles. On June 29, 2016, NMFS published a final rule in the Federal Register listing Nassau grouper as threatened under the Endangered Species Act, effective July 29, 2016 (81 FR 42268). On April 15, 2019, NMFS published a final rule to list the Gulf of Mexico Bryde’s whale as endangered, effective May 15, 2019 (84 FR 15446). On January 30, 2018, NMFS published a final rule to list the oceanic whitetip shark as threatened, effective March 1, 2018 (83 FR 4153). On January 22, 2018, NMFS published a final rule to list the giant manta ray as threatened, effective February 21, 2018 (83 FR 2916). Consequently, the ongoing consultation on the operation of the fisheries carried out under the 2006 Consolidated HMS FMP, excluding the pelagic longline fishery, was expanded to consider potential effects in light of these actions.

SERO PRD worked with the HMS Management Division, Southeast Fisheries Science Center (SEFSC), Northeast Fisheries Science Center (NEFSC), Greater Atlantic Regional Fisheries Office (GARFO), the Fisheries Statistics Division, and SERO SFD, from winter of 2016 through spring of 2018, to clarify information and data analyses on potential interactions and effects from the proposed action on species listed under the ESA and then-proposed for listing (i.e., to obtain the information necessary for a complete initiation package).

The consultation package was considered complete on May 4, 2018.

2.0 Description of the Proposed Action and Action Area

The Magnuson-Stevens Fishery Conservation and Management Act (MSA) grants authority to the Secretary of Commerce (“Secretary”) to manage HMS, i.e., tunas, swordfish, billfish, and sharks within the U.S. Exclusive Economic Zone (EEZ) in the Atlantic Ocean, Gulf of Mexico and Caribbean Sea. See 16 U.S.C. §§ 1852(a)(3) and 1802(21). The Secretary delegated that authority to the National Oceanic and Atmospheric Administration, which in turn delegated it to NMFS. The HMS Management Division within NMFS administers the Act with respect to HMS fisheries. NMFS must rebuild overfished fisheries and prevent overfishing while achieving optimum yield on a continuing basis, consistent with the National Standards and other MSA requirements. Additionally, any management measures must be consistent with other domestic laws including, but not limited to, the National Environmental Policy Act, ESA, Marine Mammal Protection Act (MMPA), and Coastal Zone Management Act. Atlantic HMS are also managed under authority of the Atlantic Tunas Convention Act (ATCA), which authorizes the Secretary of Commerce to promulgate regulations, as may be necessary and appropriate, to carry out recommendations of the International Commission for the Conservation of Atlantic Tunas (ICCAT). ICCAT is a regional fishery management organization with 52 members, including the United States. The United States helps develop recommendations aimed at promoting the conservation, management, and rebuilding of Atlantic highly migratory fish stocks, including those important to U.S. interests. ICCAT also undertakes work on management and data compilation of bycatch that are caught by fleets participating in ICCAT fisheries.

Within NMFS, the HMS Management Division has the lead in developing regulations for all Atlantic HMS fisheries, although some actions (e.g., implementation of the Atlantic Large Whale Take Reduction Plan) are taken by or in cooperation with other offices if the main legislation (e.g., the MMPA) driving the action is not the MSA or ATCA. The HMS Management Division manages Atlantic HMS fisheries in U.S. Atlantic waters including the Gulf of Mexico and Caribbean Sea. Tuna, sharks, swordfish, and billfish live throughout the Atlantic Ocean and Gulf of Mexico and often migrate long distances. Because these species cross national and international management boundaries, the HMS Management Division is responsible for managing the fisheries under the MSA and ATCA. With advice from an Advisory Panel, the HMS Management Division develops and implements conservation and management measures for Atlantic HMS species that are in need of conservation and management, taking into account all domestic and international requirements under applicable statutes including the MSA, ATCA, MMPA, ESA, and Migratory Bird Treaty Act.

Descriptions of the current regulations and management measures for the northwest Atlantic, Gulf of Mexico, and Caribbean Region HMS fisheries (excluding the pelagic longline fishery) are provided Section 2.1 through 2.9. For more information on the Atlantic HMS regulations, please see 50 CFR Part 635. For more information on HMS landings data, please see the HMS Stock Assessment and Fisheries Evaluation (SAFE) Reports at <https://www.fisheries.noaa.gov/atlantic-highly-migratory-species/atlantic-highly-migratory-species-stock-assessment-and-fisheries-evaluation-reports>.

2.1 Overview of Management Measures for Atlantic HMS

The 2006 Consolidated HMS FMP, as amended, addresses fishery management measures within federal waters of the U.S. EEZ (Figures 2.1 and 2.2). In some cases, such as management of Atlantic tunas, the 2006 Consolidated HMS FMP establishes regulations that are applicable to shore with some limited exceptions (50 CFR 635.1).

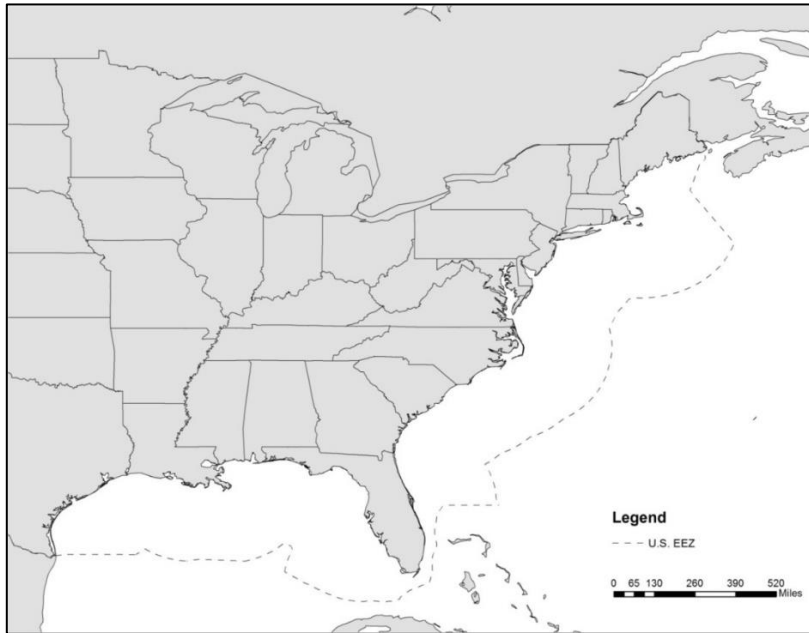


Figure 2.1 Continental 2006 Consolidated HMS FMP management area as bounded by the U.S. EEZ

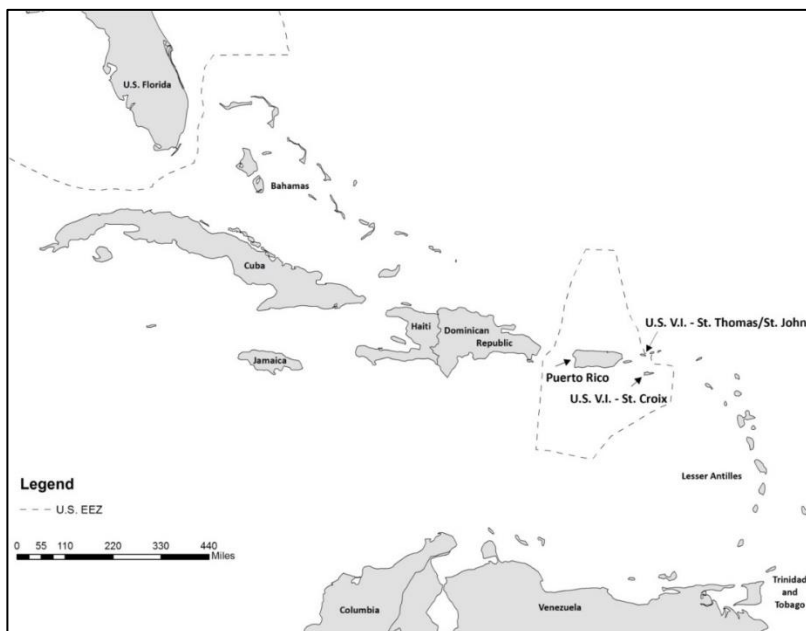


Figure 2.2 2006 Consolidated HMS FMP management areas in the Caribbean as bounded by the U.S. EEZ around Puerto Rico and the U.S. Virgin Islands

Species Managed Under the Consolidated FMP and its Amendments

NMFS manages five species of tuna under the 2006 Consolidated HMS FMP and its amendments: skipjack tuna, albacore tuna, yellowfin tuna, bigeye tuna, and bluefin tuna. Bigeye, northern albacore, yellowfin, and skipjack tunas are collectively referred to as BAYS tunas. NMFS also manages swordfish, sailfish, white marlin, blue marlin, roundscale spearfish, and longbill spearfish under the FMP. Additionally, NMFS manages 42 species of Atlantic sharks, divided into five primary groups for management: large coastal sharks (LCS), small coastal sharks (SCS), pelagic sharks, smoothhound sharks, and prohibited species. The LCS complex is comprised of 11 species including sandbar, silky, tiger, blacktip, spinner, bull, lemon, nurse, scalloped hammerhead, great hammerhead, and smooth hammerhead sharks. SCS consist of finetooth, Atlantic sharpnose, blacknose, and bonnethead sharks. Pelagic sharks consist of blue, oceanic whitetip, porbeagle, shortfin mako, and common thresher sharks. The smoothhound complex includes smooth dogfish, Gulf smoothhounds, and Florida smoothhounds. Prohibited sharks consist of sand tiger, bigeye sand tiger, whale, basking, white, dusky, bignose, Galapagos, night, Caribbean reef, smalltail, Caribbean sharpnose, narrowtooth, Atlantic angel, longfin mako, bigeye thresher, sevengill, sixgill, and bigeye sixgill sharks. The quotas for some HMS managed species are split into fishing regions. Please see the HMS SAFE Reports at <https://www.fisheries.noaa.gov/atlantic-highly-migratory-species/atlantic-highly-migratory-species-stock-assessment-and-fisheries-evaluation-reports> for more information on these species and their status.

History of 2006 Consolidated HMS FMP Amendments

Over the years, NMFS has implemented numerous amendments to the 2006 Consolidated HMS FMP, some of which affect all HMS species (e.g., Amendment 1 in 2009 and Amendment 10 in 2017, which address EFH) and other that affect specific species. Many of these amendments were undertaken to rebuild overfished stocks and to prevent or end overfishing of Atlantic sharks in commercial and recreational fisheries. Section 3.1.1 of Final Amendment 3 (2010) to the 2006 Consolidated HMS FMP includes a detailed history of domestic shark management. In addition to Amendment 3, other FMP amendments have addressed shark management, including Amendment 2 (2008, sandbar, dusky, porbeagle, and blacktip sharks); Amendment 4 (2012, Caribbean HMS measures); Amendment 5a (2013, sandbar, scalloped hammerhead, blacknose, and blacktip sharks); Amendment 6 (2015, small coastal sharks and changes to regions); Amendment 9 (2015, smoothhound sharks); Amendment 5b (2017, dusky sharks), and Amendment 11 (2019, shortfin mako sharks). Changes in management measures and regulations have generally resulted from new stock assessments, some of which have continued to find at least some shark stocks overfished, slower to rebuild than expected, or experiencing overfishing, and some of which have found the species are not overfished or are not experiencing overfishing. Some of the regulations implemented in these FMP Amendments have also been implemented to minimize the impacts of the shark fisheries on MMPA and ESA-listed species, most recently Amendment 9, which implemented the terms and conditions of the 2012 Opinion in the shark gillnet fisheries. Other amendments have addressed species other than sharks, including Amendment 8 (2013, swordfish); and Amendment 7 (2014, bluefin tuna).

For a list of complete amendments to the current Atlantic HMS FMPs, please see <https://www.fisheries.noaa.gov/atlantic-highly-migratory-species/atlantic-hms-fishery-management-plans-and-amendments>.

In addition to FMP Amendments, other regulatory actions that have been taken over the years include opening and closing of fisheries and adjustments to quota allocations.

2.2 Authorized Commercial and Recreational Gear

The gear type authorized for an activity depends upon three things: (1) the type of fishing being conducted (commercial, recreational, or scientific research); (2) the species being targeted; and (3) the type of permit which is being used for that activity. The tables below reflect which gear types may be used for which species, and additional information is provided in the appropriate sections in the HMS compliance guides³. Gear types for scientific research (which can be authorized by EFPs, scientific research permits (SRPs),⁴ display permits, and shark research fishery permits,⁵ see 50 CFR 635.32) can vary from the traditional commercial and recreational gears (e.g., include plankton nets) but are generally similar (e.g., rod and reel or bottom longline). NOTE: A vessel using or having onboard any unauthorized gear may not possess any Atlantic HMS.

Table 2.1 Authorized Commercial and Recreational Gear Types

Gear Type	Sharks	Bluefin Tuna	BAYS Tunas	Swordfish
Bandit	X	X	X	X
Bottom Longline	X			
Buoy Gear*			X**	X
Gillnet	X			
Green-stick		X	X	X
Handline	X	X	X	X
Harpoon***		X	X	X
Purse Seine		X	X	
Rod and Reel	X	X	X	X
Speargun****			X	

* Must have Swordfish Directed limited access, Swordfish Handgear limited access, or HMS Commercial Caribbean Small Boat permit.

** HMS Commercial Caribbean Small Boat permit holders only.

*** Not authorized for Charter/Headboat permit holders.

**** For use by Charter/Headboat permit holders for recreational fishing only (speared BAYS tunas may not be sold).

2.3 Commercial Fishing – Atlantic Tunas and Swordfish Fisheries

Atlantic HMS that can be landed for commercial purposes include certain tunas, swordfish, and sharks. This section addresses tunas and swordfish fishing and gear types and shark landings as

³ Atlantic HMS Fishery Compliance Guides are designed to provide a plain language summary of HMS regulations; however, they are not a substitute for the regulations found in the Code of Federal Regulations (50 CFR 635). HMS compliance guides can be found at: <https://www.fisheries.noaa.gov/atlantic-highly-migratory-species/atlantic-highly-migratory-species-fishery-compliance-guides>.

⁴ SRPs are required for scientific research activities concerning all species covered under 50 CFR part 635 regulated under the authority of the Atlantic Tunas Convention Act. 50 CFR 653.32(b).

⁵ As described in Sections 2.4.1 and 2.7, below, NMFS issues permits for participation in the shark research fishery as exempted fishing permits. 50 CFR 635.32(f). Although the shark research fishery is not restricted to using bottom longline gear, all participants to date have fished exclusively with bottom longline gear.

bycatch when targeting tunas and swordfish. Directed commercial shark fishing activities are discussed below.

2.3.1 Green-Stick

Green-stick gear may be used to harvest BAYS tunas and bluefin tuna aboard Atlantic tunas General category, HMS Charter/Headboat, and Atlantic tunas Longline permitted vessels (73 FR 54271, 50 CFR 635.21(i)). In August 2013, Amendment 8 to the Consolidated HMS FMP (78 FR 52011) also allowed green-stick gear to be used to harvest swordfish under the Swordfish General Commercial permit. This permit allows for similar gear as the Atlantic tunas General category permit to be used to harvest swordfish. The “commercial” configuration of green-stick gear generally consists of a 10.7 - 13.7 m (35-45 feet) fiberglass pole mounted to the vessel (NMFS 2014e). A heavy mainline (800-1,000-pound test line) housed in a spool is hoisted by a tether-rope mounted to the top of the pole (NMFS 2014e). The mainline is attached to a vessel and elevated or suspended above the surface of the water with no more than 10 hooks or gangions attached to the mainline (73 FR 54271). The mainline is connected to the tether-rope with a cotton breakaway cord (NMFS 2014e). At the end of the mainline, a floating decoy is attached (73 FR 54271). This decoy provides drag as the vessel moves forward and puts tension on the mainline (73 FR 54271). Several leaders hang down from the mainline at regularly spaced intervals and suspend baits so that they brush across the top of the water (Figure 2.3). As this gear is towed, the baits attached to the mainline skip across the water’s surface and flex in the fiberglass pole produces a “jigging” action that attracts fish (73 FR 54271). This gear was designed so that the mainline breaks away from the tether rope when one or more fish are hooked. Fish are hooked as they strike the baits, which most frequently results in hooking locations in the jaw or mouth area and does not often result in deep-hooking (73 FR 54271). The mainline and all the fish are then retrieved together using the spool (Wescott, 1996). The suspended line, attached gangions and/or hooks, and catch may be retrieved collectively by hand or mechanical means (73 FR 54271).

Green-stick does not constitute a pelagic longline (PLL) or a bottom longline (BLL) as defined at § 635.21(c) or § 635.21(d), respectively. Green-stick gear is also distinguished from PLL and BLL gear in that green-stick gear is actively trolled and does not have floats capable of supporting the mainline, as with PLL, nor weights and/or anchors capable of maintaining contact between the mainline and the ocean bottom, as with BLL. Green-stick can be used by Atlantic Tunas Longline category permitted vessels at times and in areas including, but not limited to, times and areas closed to longline fishing if the requirements for removal of any one of the elements of a PLL are met (73 FR 54721).

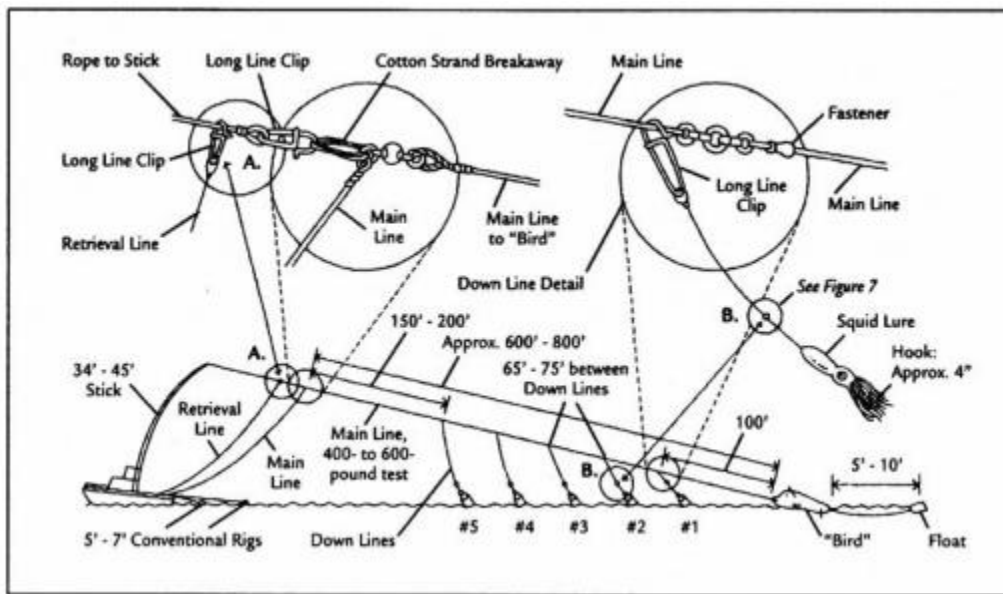


Figure 2.3 A Diagram of the Commercial Configuration of Green-stick Fishing Gear. Source: Wescott, 1996

Onboard Atlantic tunas Longline category permitted vessels, up to 20 J-hooks may be possessed for use with green-stick gear and no more than 10 J-hooks may be used with a single green-stick gear. J-hooks may not be used with PLL gear and no J-hooks may be possessed onboard a PLL vessel unless green-stick gear is also onboard. J-hooks possessed and used onboard PLL vessels may be no smaller than 1.5 inch (38.1 mm) when measured in a straight line over the longest distance from the eye to any other part of the hook (50 CFR 635.21(c)(2)(vii)(A); 50 CFR 635.21(c)(5)(iii)(B)(3)).

NMFS previously determined that its proposed action of authorizing green-stick gear for the harvest of Atlantic tunas was not likely to adversely affect ESA-listed species (2008 Memorandum from Roy E. Crabtree, PhD, to Alan D. Risenhoover). The green-stick fishery is classified as Category III under the MMPA (84 FR 22051, May 16, 2019), meaning that these fisheries have a remote likelihood of incidental mortality or serious injury to marine mammals.

Recent Catch and Landings

Determining historical landings for green-stick gear is not easily quantifiable due to the lack of reporting mechanisms for the gear type available in some fisheries data collection programs in the past (NMFS 2017). Limited data allowed the catch to be characterized and presented in the 2008 SAFE Report (NMFS 2017). In 2008, a green-stick gear code was designated for use in existing reporting systems, such as trip tickets in the southeast and electronic reporting programs in the northeast (NMFS 2017). NMFS encouraged states to utilize the green-stick gear code in their trip ticket programs to improve data on landings (NMFS 2017). Beginning in 2013, the HMS eDealer electronic reporting system was required to be used by Atlantic HMS dealers, improving the precision of green-stick landings data (NMFS 2017).

Table 2.2 Select Landings with Greenstick Gear (lb. ww) in 2013-2017

Species	Region	2013	2014	2015	2016	2017
Yellowfin tuna	Atlantic	43,175	57,064	44,673	35334	77753
	Gulf of Mexico	19,212	1,082	-	1,055	10540
Bigeye tuna	Atlantic	-	-	-	1,666	-
	Gulf of Mexico	-	-	-	-	-

Note: Additional landings of other species with greenstick gear have occurred, but given the limited number of vessels reporting such landings, this information cannot be displayed due to MSA confidentiality requirements.

Source: Atlantic HMS Electronic Dealer Reporting System

NMFS and the Louisiana Department of Wildlife and Fisheries investigated the catch and bycatch of green-stick gear during 2012-2016 in the northern GOM through a study funded by the NOAA Bycatch Reduction Engineering Program. The final report from that study is available on request from the HMS Management Division.

2.3.2 Purse Seine

Purse seine gear may be used to harvest bluefin tuna and BAYS tunas. Purse seine gear consists of a floated and weighted encircling net that is closed by means of a drawstring, known as a purseline, threaded through rings attached to the bottom of the net. Atlantic tuna purse seining operations typically have used spotter aircraft to locate fish schools. Once a school is spotted, a vessel, with the aid of a smaller skiff, intercepts and uses the large net to encircle it. Once encircled, the purseline is pulled, closing the bottom of the net and preventing escape. The net is hauled back onboard using a powerblock, and the tunas are removed and placed onboard the larger vessel. A purse seine used in directed fishing for bluefin tuna must have a mesh size equal to or smaller than 4.5 inches (11.4 cm) in the main body (stretched when wet) and must have at least 24-count thread throughout the net (50 CFR 635.21(e)(1)). Vessels participating in the Atlantic tunas purse seine fishery are required to target the larger size class bluefin tuna, more specifically the giant size class (≥ 81 inches) and are granted a tolerance limit for large medium size class bluefin tuna (73 to < 81 inches) (i.e., large medium catch may not exceed 15% by weight of the total amount of giant bluefin tuna landed during a season) (50 CFR 635.23(e)).

Vessels using purse seine nets have participated in the U.S. Atlantic tuna fishery as early as the 1930s, although the level of activity escalated in targeting and land bluefin off the coast of Gloucester, MA until the 1950s. In 1958, commercial purse seining effort for Atlantic tunas began with a single vessel in Cape Cod Bay and expanded rapidly into the region between Cape Hatteras and Cape Cod during the early 1960s. Since the 1970s, purse seine vessels focused their effort on giant bluefin, versus other tunas, due to the emerging international market that developed for giant bluefin in the late 1970s. These fresh caught bluefin were primarily flown directly to Japan for processing into sushi or sashimi. A limited entry permit system with non-transferable individual vessel quotas for purse seining was established in 1982, effectively excluding any new entrants into this category. Equal baseline quotas of bluefin were assigned to individual vessels by regulation; the individual vessel quota system was possible given the small pool of ownership in this sector of the fishery, i.e., five qualified participants. Purse seine landings historically have made up approximately 20 percent of the total annual U.S. landings of bluefin tuna, but there has been no, to little, activity from this segment of the fishery for a number of years (NMFS 2014e).

The baseline Purse Seine category quota currently is codified as 219.5 mt or 18.6% of the U.S. quota (50 CFR 635.27(a)). Annually, NMFS makes determinations regarding the start of the purse seine fishery based on variations in seasonal distribution, abundance or migratory patterns of bluefin tuna, cumulative and projected landings, the potential for gear conflicts on the fishing grounds, and market impacts. NMFS also makes determinations regarding quota allocations to each participant, applying a formula adopted in Amendment 7 to the 2006 Consolidated HMS FMP (50 CFR 635.27(a)(4)). In the scoping document for Amendment 13 to the 2006 Consolidated HMS FMP, which addresses management of bluefin tuna, NMFS includes elimination of the Purse Seine category among potential management options to consider in the future because there have been no landings of bluefin tuna in this category since 2015 (84 FR 23020, May 21, 2019).

Vessel Monitoring System Requirements

Vessels with an Atlantic Tunas Purse Seine category permit must have a Vessel Monitoring System (VMS) unit installed on their vessel in order to use purse seine gear. The VMS unit must submit automatic position reports every hour, 24 hours a day, unless a valid power down exemption has been granted by NMFS law enforcement. 50 CFR 635.69(e)(1). Vessels fishing with purse seine gear must submit a “Highly Migratory Species Bluefin Tuna Catch Report” through VMS within 12 hours of completion of each purse seine set. The report must include: date the set was made; area in which the set was made; and the length of all bluefin tuna retained (actual) and discarded dead or released alive (approximate), including reporting of zero bluefin on a set. 50 CFR 635.69(e)(4)(ii).

Atlantic Tunas Purse Seine Fishery Observer Coverage

ICCAT Recommendation 10-10, Recommendation by ICCAT to Establish Minimum Standards for Fishing Vessel Scientific Observer Programs, required a minimum of 5% observer coverage of fishing effort in the purse seine fishery, as measured in number of sets or trips. NMFS implemented the requirement in 2011.

Recent Catch and Landings

Table 2.3 shows purse seine catch (landings + dead discards) of Atlantic bluefin tuna from 2008 through 2017. No other tuna species were landed by vessels permitted in the Purse Seine category during this time; purse seine fishing effort has been directed only on bluefin tuna. Purse seine landings historically made up approximately 20 percent of the total annual U.S. landings of bluefin tuna (about 25 percent of total commercial landings), but over the past 20 years have only accounted for a small percentage of landings (NMFS 2017). There have been no landings in the fishery since 2015.

Table 2.3 Domestic Atlantic Bluefin Tuna Catch (mt ww) for the Purse Seine Fishery in the Northwest Atlantic Fishing Area (2008-2017)

Species	2009	2010	2011	2012	2013	2014	2015	2016	2017
Bluefin tuna	11.4	0.0	0.0	1.7	42.5	41.8	38.8	0.0	0.0

Source: NMFS 2019

In 2016, 2017, and 2018 NMFS did not open (i.e., announce a start date for) the Atlantic tunas purse seine fishery because there were no active vessels permitted to fish for bluefin tuna with purse seine gear and therefore there was no catch of bluefin tuna in 2016, 2017, and 2018 (NMFS 2019). Although NMFS received an EFP application for purse seine fishing (similar to those submitted for 2014 and 2015), NMFS did not grant the EFP (NMFS 2017).

2.3.3 Commercial Handgear

Commercial handgears, including handline, harpoon, rod and reel, buoy gear and bandit gear, are used to fish for Atlantic HMS on private vessels, charter vessels, and headboat vessels. Rod and reel gear may be deployed from a vessel that is anchored, drifting, or underway (trolling). In general, trolling consists of dragging baits or lures through, on top of, or even above the water's surface. While trolling, vessels often use outriggers to assist in spreading out or elevating baits or lures and to prevent fishing lines from tangling.

The handgear fisheries for all HMS are typically most active during the summer and fall, although in the Mid-Atlantic and Gulf of Mexico, fishing with handgear occurs during the winter months. Fishing usually takes place between a few and two hundred kilometers (km) from shore and, for those vessels using bait, the baitfish typically includes herring, mackerel, whiting, mullet, menhaden, ballyhoo, butterfish, and squid.

The majority of bluefin landings are by handgear fisheries in the commercial Atlantic tunas General category and recreational HMS Angling and HMS Charter/Headboat categories. Vessels permitted in the Atlantic tunas General category are focused in New England during the summer and fall and the South Atlantic during the winter. These vessels tend to fish in offshore, deeper waters.

The commercial handgear fishery for bluefin tuna occurs in New England, and off the coast of southern Atlantic states, such as Virginia, North Carolina, and South Carolina, with vessels targeting large medium and giant bluefin tuna. Bluefin tuna commercial landings are the predominate handgear landings, in metric tons (mt) by geographic region: Gulf of Mexico, South Atlantic, Mid-Atlantic, and Northeast (the South Atlantic region ends at Cape Hatteras, and the Mid-Atlantic region ends at eastern Long Island, New York).

Commercial landings declined during the early 2000s, but have increased over the past five years. All commercial landings, regardless of year, have been within the overall U.S. annual quotas as authorized at ICCAT and implemented domestically by regulation. Targeting bluefin tuna in the Gulf of Mexico, the known spawning grounds for the western Atlantic stock, is prohibited, although some incidental harvest is allowed. The majority of U.S. commercial handgear fishing activities for bigeye, albacore, yellowfin, and skipjack tunas take place along the east coast of the United States. Beyond these general patterns, the availability of Atlantic tunas at a specific location and time is highly dependent on environmental variables that fluctuate from year to year.

The U.S. Atlantic tuna commercial handgear fisheries are currently managed through an open access vessel permit program. Vessels that wish to sell their Atlantic tunas must obtain a permit in one of the following categories: General (authorizes handgear including rod and reel, harpoon,

handline, bandit gear, and green-stick), Harpoon (authorizes harpoon only), or Charter/Headboat (authorizes for-hire passengers to recreationally fish for any HMS species with rod and reel, for tunas, sharks, or swordfish with handline, for tunas with bandit gear and green-stick, and free-swimming tunas (excluding Bluefin) with a speargun) (for more detailed permit descriptions see <https://hmspermits.noaa.gov/>). These federally-permitted vessels may also need permits from the states they operate from in order to land and sell their catch, and are encouraged to check with their local state fishery management agency regarding these requirements. Federally-permitted vessels are required to meet all applicable U.S. Coast Guard safety gear requirements as well as sell their Atlantic tunas only to federally-permitted Atlantic tunas dealers.

The Commercial Caribbean Small Boat permit is open access and valid in the U.S. Caribbean region on vessels that are less than 45 feet long. This permit allows the commercial retention of tunas, swordfish, and sharks when using handgear (handline, buoy gear, harpoon, rod and reel, or bandit gear). The current retention limit for bigeye, northern albacore, yellowfin, and skipjack tunas (collectively referred to as BAYS tuna) is 10 fish, and the retention limit for North Atlantic swordfish is two fish. The shark retention limit is zero; however, if the retention limit were increased, permit holders would be allowed to retain and sell non-prohibited species of sharks.

The Swordfish General Commercial permit is open access and can be held in conjunction with the Atlantic Tunas Harpoon and General category permits. Permit holders can only use rod and reel, handline, bandit gear, green-stick, or harpoon gear. The swordfish retention limit under this permit may be set between zero and six fish per vessel per trip. The default retention limits for North Atlantic swordfish are three in the northwest Atlantic and Gulf of Mexico, two in the U.S. Caribbean, and zero in the Florida Swordfish Management Area.

Table 2.4 displays the estimated number of rod and reel and handline trips targeting large pelagic species (e.g., tunas, billfishes, swordfish, sharks, wahoo, dolphin, and amberjack) from Maine through Virginia from 2012 to 2017. The trips include both commercial and recreational trips, and are not specific to any particular species.

Table 2.4 Estimated Number of Rod and Reel and Handline Trips Targeting Atlantic Large Pelagic Species, by State (ME-VA 2012-2017)

Year	Area							Total
	NH/ME	MA	CT/RI	NY	NJ (North)	NJ (South) and MD/DE	VA	
Private Vessels								
2012	8,408	19,096	6,189	6,425	5,447	13,682	2,445	61,692
2013	7,100	12,883	2,366	6,648	4,104	11,519	2,187	46,807
2014	4,289	12,758	3,639	6,777	4,589	11,575	1,972	45,559
2015	4,074	12,130	3,336	7,068	3,166	11,741	2,522	44,037
2016	4,224	10,511	3,802	6,481	3,337	11,193	2,754	42,302
2017	5397	12088	2909	9060	3843	10316	2082	45695
Charter Vessels								
2012	1,570	4,248	465	1,211	1,437	2,910	619	12,462
2013	868	3,181	999	1,010	1,113	2,763	399	10,333
2014	836	3,294	592	1,220	1,199	2,172	345	9,658
2015	1,262	3,835	613	1,458	1,167	1,730	499	10,572

2016	669	3,756	552	1,423	1,439	2,798	263	10,900
2017	998	3934	329	1866	1554	2657	822	12160

Source: Large Pelagics Survey (LPS), NMFS 2018

The commercial North Atlantic swordfish fishery began in the early 1800s as a harpoon fishery off the New England coast. Sailing vessels used harpoons to capture swordfish on extended trips to the Hudson Canyon and Georges Bank during summer months. For more than 150 years, up until the 1960s, most U.S. commercial swordfish were captured using harpoons or handlines. A small U.S. recreational swordfish fishery developed in the 1920s using rod and reel and handline, primarily from Massachusetts to New York. As diesel engines came to replace sail, PLL gear eventually replaced harpoons as the primary commercial swordfish gear during the 1960s. As the swordfish stock has rebuilt over the past decade, more fish have recruited to larger sizes and the range of fish captured on traditional handgears has expanded. Rod and reel and harpoon gears have recently become more economically viable again in more areas, including New England and the Gulf of Mexico. A commercial swordfish fishery utilizing handgear (especially buoy gear) exists primarily off the east coast of Florida, but also occurs in other locations of the Atlantic, Gulf of Mexico, and U.S. Caribbean. The handgear fishery for swordfish is currently managed through a mix of open access and limited access vessel permits. The location of Swordfish Handgear limited access permits has shifted south over the last decade. In 2004, the majority of the permits were located in Rhode Island (28 permits), Florida (20 permits), and Massachusetts (17 permits). Between 2004 and 2018, the number of Swordfish Handgear limited access permits in Florida more than doubled from 20 to 52 permits (suggesting that this is an important location for this fishery). During this same timeframe, the number of permits in Rhode Island decreased to 12 and in Massachusetts to 7. For updated permit information see <https://www.fisheries.noaa.gov/southeast/frequent-freedom-information-act-requests-southeast-region>.

Buoy gear is a fishing gear consisting of one or more floatation devices supporting a single mainline to which no more than two hooks or gangions are attached. The only permits that authorize the use of buoy gear are the Swordfish Handgear limited access permit, the Swordfish Directed limited access permit (only when held in combination with a shark limited access permit and a Tunas Longline category permit), and the Commercial Caribbean Small Boat permit (which is only valid in the U.S Caribbean territories of Puerto Rico and U.S. Virgin Islands). Buoy gear is generally used to target swordfish and is usually fished at night. Authorized permit holders may not possess or deploy more than 35 floatation devices and may not deploy more than 35 individual buoy gears per vessel. Buoy gear must be constructed and deployed so that the hooks or gangions or both are attached to the vertical portion of the mainline. Floatation devices may be attached to one, but not both ends of the mainline, and no hooks or gangions may be attached to any floatation device or horizontal portion of the mainline. If more than one floatation device is attached to a buoy gear, no hook or gangion may be attached to the mainline between them. Individual buoy gears may not be linked, clipped, or connected together in any way. Buoy gears must be released and retrieved by hand. All deployed buoy gear must have some type of monitoring equipment affixed to it including, but not limited to, radar reflectors, beeper devices, lights, or reflective tape. If only reflective tape is affixed, the vessel deploying the buoy gear must possess on board an operable spotlight capable of illuminating deployed floatation devices. If a gear monitoring device is positively buoyant, and rigged to be attached to

a fishing gear, it is included in the floatation device vessel limit and must be marked appropriately.

Buoy gear effort and catch data are available in HMS SAFE Reports from 2007 through 2017. Prior to 2007, buoy gear catch data were included in handline catch data. In the Caribbean, buoy gear (referred to in the Caribbean as yo yo gear) is used to target swordfish and tunas and may have incidental catches of sharks.

Buoy gear effort, as reported by the fishery, and published in the most recent HMS SAFE Report is presented from 2012 to 2017 in Table 2.5.

Table 2.5 Reported Buoy Gear Effort (2012-2017)

Specifications	2012	2013	2014	2015	2016	2017
Number of Vessels	55	46	39	37	42	36
Number of Trips	688	629	467	353	337	252
Average buoy gears deployed per trip	14.1	17.95	20.9	21.1	23.6	23.4
Total Number of Set Hooks	11,639	12,557	10,740	8,267	8,588	6282
Average Number of Hooks per gear	1.2	1.1	1.1	1.1	1.1	1.1

Source: UDP, NMFS 2018

2.4 Commercial Fishing - Directed Shark Fishery

The HMS Management Division currently manages sharks in five management units: LCS, SCS, pelagic sharks, the smoothhound complex, and prohibited species. Prior to the implementation of Amendment 2 to the 2006 Consolidated HMS FMP in 2008, the primary target species in the fisheries of the Atlantic and Gulf of Mexico coasts were sandbar and blacktip sharks, although many other shark species were caught as well. Since Amendment 2, which significantly reduced the sandbar quota to only a small research fishery, the fishermen in these areas primarily target blacktip and Atlantic sharpnose sharks. The majority of participants in the shark fisheries off the Caribbean are small-scale commercial vessels using handgear (handline, rod and reel). A summary of commercial compliance regulations is available in the HMS Commercial Compliance Guide found at <https://www.fisheries.noaa.gov/atlantic-highly-migratory-species/atlantic-highly-migratory-species-fishery-compliance-guides>.

2.4.1 Bottom Longline

Bottom longline gear is the primary commercial gear employed for targeting LCS in all regions. The commercial shark bottom longline fishery is active in the U.S. Atlantic Ocean from Virginia to Florida and throughout the Gulf of Mexico. Vessels in this fishery primarily target large coastal shark species, e.g., sandbar and blacktip sharks (Hale and Carlson 2007; Morgan et al. 2009).

Longline characteristics vary regionally, with gear normally consisting of 8–24 km of longline and 500–1500 hooks (Hale and Carlson 2007; Morgan et al. 2009). Gear is generally set at sunset, allowed to soak overnight before hauling back in the morning (Hale and Carlson 2007; Morgan et al. 2009). Fishermen targeting sharks with bottom longline gear are opportunistic and

often maintain permits for Fishery Council-managed fisheries such as reef fish, snapper/grouper, tilefish, and other teleosts. Minor modifications to how and where the gear is deployed allow fishermen to harvest sharks and teleosts on the same trip. Seasons, quota availability, market prices, and other factors influence decisions concerning whether to target sharks, teleosts, or both on a given trip. The gear typically consists of a heavy monofilament mainline with lighter weight monofilament gangions. Some fishermen may occasionally use a flexible 1/16 inch wire rope as gangion material or as a short leader above the hook (Hale et al. 2010).

Several exempted fishing permit recipients targeting sharks, as well as several entities possessing letters acknowledging their activities as scientific research conducted from scientific research vessels, have been using a modified bottom longline gear called drumline gear. Drumline consists of a single float with a 700 lb. monofilament mainline that is weighted to maintain contact with the bottom. Up to 20 hooks are typically used on the drumline gear. This gear typically has short soak times between one and two hours, which, maximizes shark survivability and minimizes bycatch.

The commercial shark bottom longline fishery has been the subject of a number of management measures since 1993 and fishermen commonly switch tactics to reflect these changes in an attempt to maintain yield. Current commercial regulations include limited access vessel permits requirements, commercial quotas, vessel retention limits, a prohibition on landing 20 species of sharks (one of these species can be landed in the shark research fishery), numerous closed areas, gear restrictions, landing restrictions (including requiring all sharks be landed with fins naturally attached), fishing regions, vessel monitoring system requirements, dealer permits, and vessel and dealer reporting requirements (Figure 2.4). Vessels that have bottom longline gear on board and that have been issued, or are required to have been issued, a directed shark limited access permit under § 635.4(e) must have only circle hooks as defined at § 635.2 on board.

A limited number of fishermen are selected each year to participate in the shark research fishery, which operates to allow NMFS opportunities to collect life history data and catch data for future stock assessments. Participants in the shark research fishery are subject to 100% observer coverage on trips (Mathers et al. 2017, NMFS 2018). Participants must fish under regulations specific to the shark research fishery (such as hook limits and bycatch caps for dusky shark). For the shark research fishery, NMFS annually publishes in the Federal Register a notice describing the expected research objectives for the following fishing year. This description may include information such as the number of vessels needed, regions and seasons for which vessels are needed, the specific criteria for selection, and the application deadline. These objectives and associated restrictions are expressed in the permit terms. Since 2012, NMFS has allowed vessels participating in the shark research fishery to harvest all non-prohibited species of sharks, including sandbar sharks. Research fishery participants' permits specify that they are required to land all catch of shark species that are legal under a directed shark permit (including sandbar shark, which is otherwise prohibited) unless they can be released alive. In 2015, HMS continued the 2012 amended model which permits one 150 hook 'feeler' set with a soak time of no more than two hours and one 300 hook set with no soak limit. Limits on the number of permitted dusky shark interactions were established by region and as permit conditions several years ago, given the species interactions within the sandbar fishery and the stock's status.

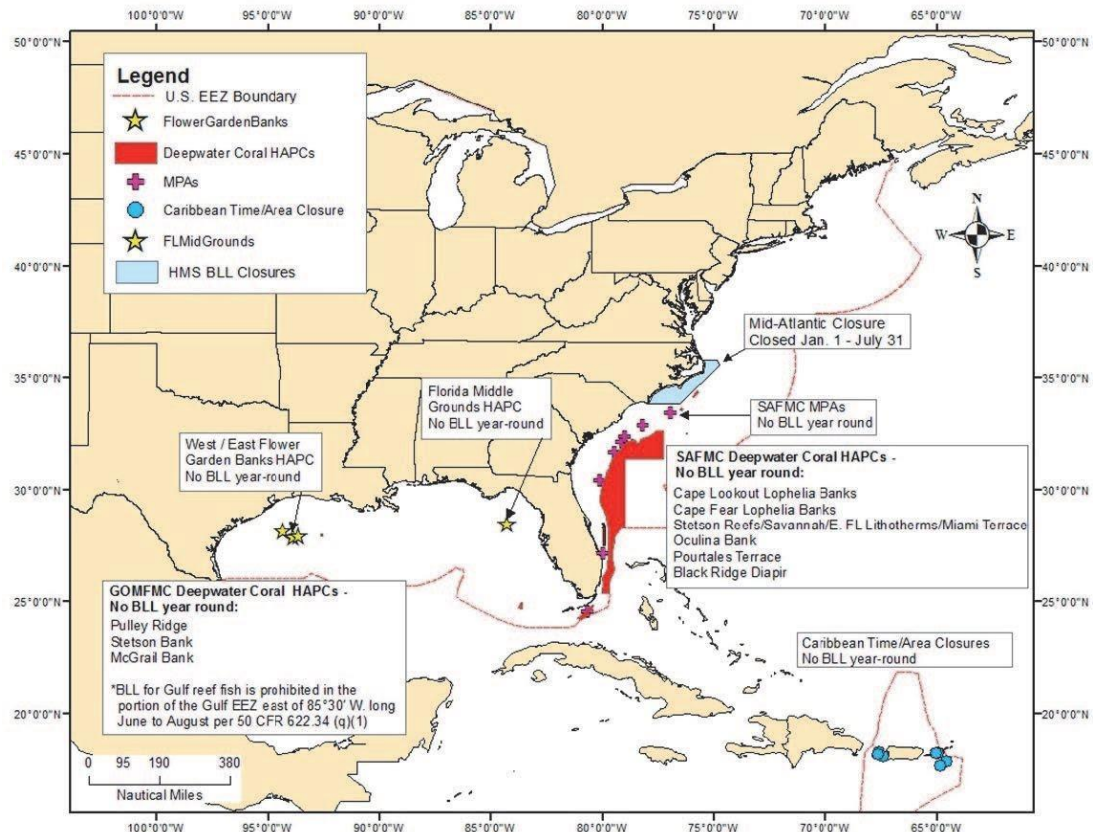


Figure 2.4 Bottom Longline Fishing Areas within the Atlantic and Season Closures Source: HMS Commercial Compliance Guide (2017)

Commercial Bottom Longline Fishery Observer Program

Since 2002, shark bottom longline vessels have been required to take a NMFS-approved scientific observer if selected. As noted earlier, participants in the shark research fishery are subject to 100% observer coverage on trips (Mathers et al. 2017, NMFS 2018). Outside the research fishery (i.e., the non-research bottom longline fishery) and depending on the time of year and fishing season, NMFS randomly selects for observer coverage vessels with current, valid directed shark permits that reported fishing with longline gear in the previous year. Target observer coverage for these vessels is 5-10% of trips (Enzenauer et al. 2016). Observer coverage in some years was subjected to limits spatially and temporally due to the availability of funding (Carlson et al. 2012).

In 2017, the bottom longline observer program observed a total of 150 bottom longline hauls (defined as setting gear, soaking gear for some duration of time, and retrieving gear) in 83 trips (defined as from the time a vessel leaves the port until the vessel returns to port and lands catch, including multiple hauls therein). Of the observed trips, 61 were taken by shark research fishery participants (total of 104 hauls) and 22 were taken outside of the shark research fishery in the southern Atlantic and Gulf of Mexico (total of 46 hauls) (Mathers et al. 2018).

Effort

In 2016, hauls targeting LCS on trips taken outside of the research fishery used bottom longline with a mainline length of 0.2 to 8.0 km (average of 3.3 km), bottom depth fished ranged from 6.1 to 880.9 m (average of 40.5 km), number of hooks deployed ranged from 25 to 509 hooks (average of 258 hooks / set), and average soak duration was 8 hours (Mathers et al. 2017). Both circle and J hooks are used; the type(s) and size of hook depends on which species is being targeted. The most commonly used hook was both the 18.0 circle hook (23.7 %) and the 9.0 J hook (23.7 %). The next commonly used hook was the 16.0 circle hook (13.2 %) followed by 9.0 and 14.0 circle hook and the 3.0 J hook (10.5 %). Hauls deployed by shark research fishery participants used bottom longline with a mainline length 2.2 to 11.0 km (average of 4.4 km), bottom depth fished ranged from 9.1 to 149.7m (average of 32.9m), number of hooks deployed ranged from 72 to 300 hooks (average of 231 hooks fished; note that there are hook limits on the shark research fishery trips), and average soak duration was 5.3 hours (Mathers et al. 2017). The most commonly used hook was the 16.0 circle hook (35.8 %) and the second most common hook was the 12.0 J hook (23.5 %). The reported bottom longline effort for fishermen targeting sharks by region from 2012 through 2016 is provided in Table 2.6. The Atlantic region has more vessels and trips targeting sharks, but the number of trips targeting sharks in the Gulf of Mexico region has surpassed the Atlantic region in 2012-2014. Distribution of trips was more evenly split between Atlantic and Gulf of Mexico regions in 2016. The number of trips is defined as targeting sharks if 75% of the landings, by weight, were sharks.

Table 2.6 Reported Bottom Longline Effort Targeting Sharks (2012-2016)

Specifications	Region	2012	2013	2014	2015	2016
Number of Vessels	Gulf of Mexico	20	16	20	18	16
	Atlantic	21	24	19	14	13
Number of Trips	Gulf of Mexico	379	457	604	527	259
	Atlantic	281	329	369	330	282
Average Sets per Trip	Gulf of Mexico	1.2	1.1	1.1	1.1	1.2
	Atlantic	1.5	1.5	1.7	1.8	1.4
Total Number of Set Hooks	Gulf of Mexico	99,675	105,559	139,709	139,956	89,123
	Atlantic	98,094	136,475	193,561	170,032	104,665
Average Number of Hooks per Set	Gulf of Mexico	229.0	212.1	206.1	236.1	272.3
	Atlantic	237.1	253.5	276.7	294.9	269.6
Total Soak Time (Hours)	Gulf of Mexico	2,912.0	2,589.5	3,011.0	2917	1,408
	Atlantic	2,289.5	2,438.0	2,649.5	2293	2,041
Average Mainline Length (Miles)	Gulf of Mexico	2.8	2.1	1.9	2.1	2.6
	Atlantic	3.9	3.4	3.4	3.8	3.6

Source: United Data Processing, NMFS 2017

2.4.2 Gillnet

Gillnet gear is the primary gear for vessels directing on small coastal and smoothhound sharks, although vessels directing on other species can also catch shark species. Vessels participating in Atlantic (including the Gulf of Mexico) shark gillnet fisheries typically possess permits for other Council and/or state managed fisheries and will deploy nets in several configurations based on target species including drift, strike, and sink gillnets. There are gillnet fisheries that occur off

the southeast U.S. coast and Gulf of Mexico regions that target small coastal sharks (referred to hereafter as the Southeast shark gillnet fishery, an HMS fishery part of the proposed action) and that target finfish (i.e., king and Spanish mackerel fisheries; these fisheries are not part of the proposed action), as well as the gillnet fisheries in the Northeast region that target smoothhounds sharks (referred to hereafter as the smoothhound gillnet fishery, an HMS fishery part of the proposed action) and that target finfish (e.g., bluefish, various groundfish; these fisheries not part of the proposed action). The majority of the vessels and trips targeting sharks with gillnets occur in the southern portion of the Atlantic region, primarily offshore of Georgia and Florida (i.e., the majority the vessels and trips targeting sharks with gillnets participate in the Southeast shark gillnet fishery). The southeast shark gillnet fishery operates mostly in inshore waters ranging from 2 - 30 m. Many of the commercial regulations for Atlantic shark fisheries are the same for both the bottom longline and gillnet fishery, including, but not limited to: seasons, quotas, species complexes, permit requirements, authorized/prohibited species, and retention limits. 50 CFR 635. Examples of regulations that are specific to all Atlantic shark gillnet fishing include: total net length regulation (2.5 km), requiring that drift gillnets remain attached to the vessel, the need to conduct net checks every two hours when drift gillnet gear is deployed (50 CFR 635.21(g)(2)), and a soak time limit of 24 hours for sink gillnets measured from the time the sink gillnet first enters the water to the time it is completely removed from the water (50 CFR Part 635.21(g)(3)).

The Atlantic Large Whale Take Reduction Plan specifies a number of restrictions on fishermen using gillnet gear, including fishermen using shark gillnet gear (defined as gillnet gear with stretched mesh greater or equal to 5 inches). Requirements in these areas include gear marking, observer coverage, and vessel monitoring systems during times when the areas are not closed to gillnets. The Southeast U.S. Restricted Area and Southeast U.S. Monitoring Area is shown in Figure 2.5. Caribbean closed areas: Fishing for HMS with gillnet gear is prohibited year-round in several distinct areas off the U.S. Virgin Islands and Puerto Rico (Figure 2.5).

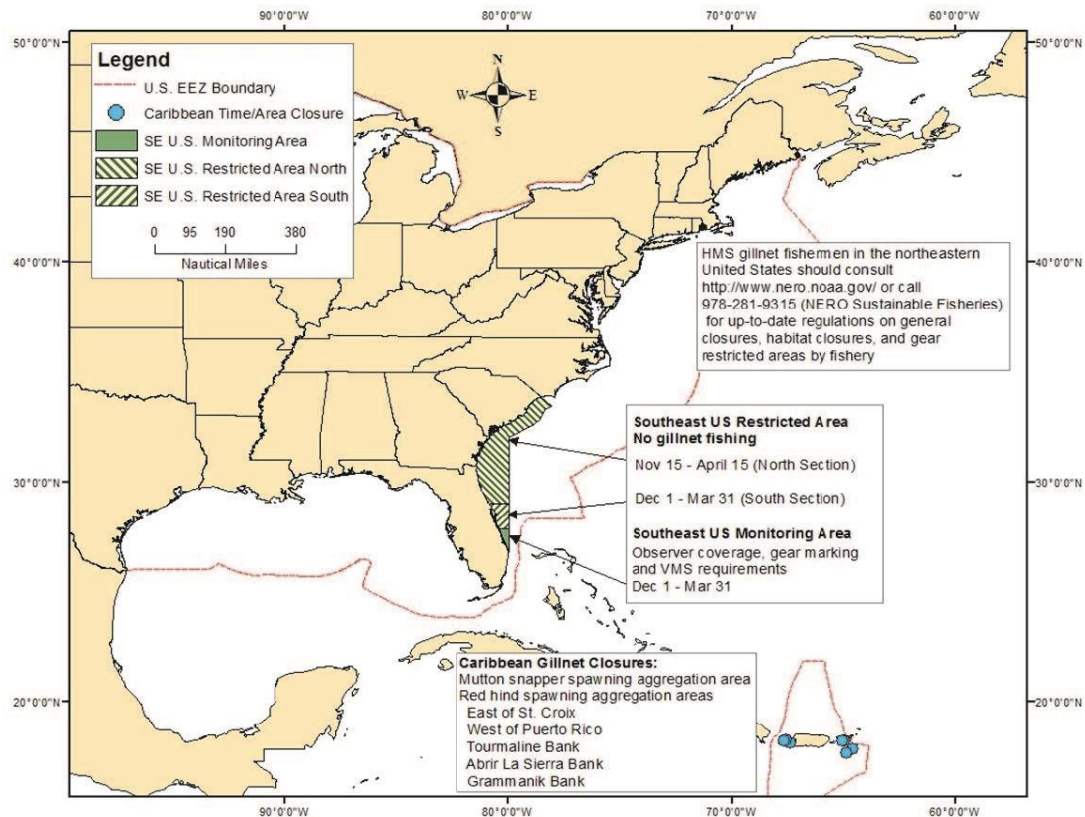


Figure 2.5. Time/Area Closures that Restrict Use of Gillnet Gear in the Atlantic Ocean, Gulf of Mexico, and Caribbean Sea

Source: HMS Commercial Compliance Guide (2017)

Commercial Gillnet Fishery Observer Program

The Shark Gillnet Observer Program (SGOP) is coordinated by SEFSC; most smoothhound shark trips in the Mid-Atlantic are observed by GARFO as part of the multispecies observer program. From 1999 through 2004, there was 100% observer coverage of the Southeast shark drift gillnet fishery during the North Atlantic right whale calving season (November 15-March 31). This coverage level was in response to a May 1997 HMS Opinion, which specified this requirement as part of a RPA to avoid jeopardy of North Atlantic right whales. The requirement was implemented via the 1999 Atlantic Large Whale Take Reduction Plan (ALWTRP) and the 1999 HMS FMP. Outside this season (April 1–November 14), the level of observer coverage had to attain a sample size large enough to provide estimates of sea turtle and smalltooth sawfish interactions with a coefficient of variation of 0.3, as recommended by NMFS (2004d). In 2005, the shark gillnet observer program was expanded to include all vessels that have an active directed shark permit and fish with sink gillnet gear. These vessels were not previously subject to observer coverage because they were either targeting non-HMS or were not fishing gillnets in a drift or strike-net fashion. Amendments to the ALWTRP regulations in 2007 vacated the 100% observer coverage requirement during North Atlantic right whale season. Observer resources were reallocated allowing all anchored (sink, stab, and set), strike, and drift gillnet vessels, from Florida to North Carolina, to be observed year-round (Baremore et al. 2007).

Vessels are randomly selected for observer coverage on a seasonal basis (winter, spring, summer, and fall) from a pool of vessels that had either a current directed or incidental shark permit and reported fishing with gillnet gear during the previous year. Permit holders selected for participating in the program are notified approximately a month before the upcoming fishing season. Upon notification, the permit holder must contact NMFS and indicate their intent to fish in the upcoming season. For each set and haulback, observers record beginning and end times of setting and hauling, estimated length of net set, sea and wind states, latitude and longitude coordinates, and water depth. Observers monitor the catch and bycatch as the nets are hauled aboard. Disposition (kept, discarded alive, or discarded dead) is recorded for each species brought on board, and measurements/samples of 10 randomly selected individuals from each species are taken if time permits (Baremore et al. 2007).

Effort

Gillnet gear is the primary gear for vessels directing on small coastal sharks, although such vessels can also catch other shark species. The data presented in this section focus on the Southeast shark gillnet fishery and the smoothhound shark gillnet fishery. The overall gillnet effort targeting sharks by region from 2012 through 2016 is shown in Table 2.7. The majority of the vessels and trips targeting sharks occur in the southern portion of the Atlantic region. Most of the data from the Gulf of Mexico region is considered confidential since fewer than three vessels used gillnet gear to target sharks in the region, and the data cannot be aggregated consistent with MSA requirements related to confidentiality of data collected under the MSA.

Table 2.7 Reported Gillnet Effort in the U.S. Atlantic and Gulf of Mexico Regions Targeting Sharks (2012-2016)

Specifications	Region	2012	2013	2014	2015	2016
Number of Vessels	Gulf of Mexico	3	C	C	C	0
	Atlantic	33	22	23	19	21
Number of Trips	Gulf of Mexico	46	C	C	C	0
	Atlantic	366	305	348	160	206
Average Sets per Trip	Gulf of Mexico	2.0	C	C	C	n/a
	Atlantic	1.5	1.1	1.0	2.1	1.8
Total Soak Time (Hours)	Gulf of Mexico	945.0	C	C	C	n/a
	Atlantic	1,074.5	849.0	1,148.5	537.8	852.5
Average Gillnet Length (Yards)	Gulf of Mexico	1443.5	C	C	C	n/a
	Atlantic	844.4	761.0	771.6	725.6	1,155.1
Average Mesh Size (Inches, Stretched Mesh)	Gulf of Mexico	7.9	C	C	C	n/a
	Atlantic	4.8	5.0	5.2	5.2	5.2

Note: Due to confidentiality requirements (C) under the Magnuson-Stevens Act, some of the data are not presented
Source: Unified Data Processing; NMFS 2017

All Atlantic HMS fishing tournaments are required to register with NMFS at least four weeks prior to the commencement of tournament fishing activities. Tournament operators may elect to register tournaments by submitting a registration form to NOAA Fisheries, or via online

registration. If selected, tournament operators are required to report the results of their tournament to the Atlantic Tournament Registration (ATR) System.

All non-tournament recreational landings of Atlantic marlins, roundscale spearfish, sailfish, bluefin tuna (including dead discards), and swordfish must also be reported to NMFS through dedicated calls lines or the Automated Landings Reporting System (ALRS) within 24 hours of landing. In Maryland and North Carolina, vessel owners are required to report their billfish bluefin tuna, and some shark landings through the submission of catch cards at state-operated landings stations. Participation in the Large Pelagics Survey (LPS) or MRIP surveys does not fulfill reporting obligations; vessel operators must still report bluefin tuna, billfish and swordfish as described above. MRIP funds and conducts various surveys and studies of recreational fishing activities and the LPS is an MRIP survey that is specific to Atlantic HMS. The LPS is conducted from Virginia to Maine during June, July, and August, and consists of dockside interviews and phone surveys to collect details on recreational fishing trips, catch, and landings.

Recreational shark landings are required to be reported to NMFS when an angler is required to participate in the LPS or MRIP. However, as of 2013 for vessel owners in Maryland, and 2014 for vessel owners in North Carolina, shark landings must be reported on catch cards at state-operated landings stations.

Bycatch can result in death or injury to discarded fish and is incorporated into fish stock assessments and into the evaluation of management measures. Bycatch in the recreational rod and reel fishery is difficult to quantify because many fishermen simply value the experience of fishing and may not be targeting a particular species. The 1999 Billfish Amendment established a catch-and-release fishery management program for the recreational Atlantic billfish fishery. Atlantic billfish that are released alive, regardless of size, are not considered bycatch, since the definition of “bycatch” under the MSA does not include fish released alive under a recreational catch and release fishery management program. 16 U.S.C. 1802(2). The recreational white shark fishery is, by regulation a catch-and-release fishery only, and white sharks similarly are not considered bycatch (CFR Title 50 Part 635.26(c)). Bycatch (dead discards) of bluefin tuna must be reported online or via phone.

On April 4, 2017, NMFS published its final rule for Amendment 5b to the 2006 Consolidated HMS FMP (82 FR 16478). The purpose of the rule was to reduce dusky shark fishing mortality as needed to end overfishing and rebuild the stock, consistent with the results of the 2016 stock assessment update to the Southeast Data and Assessment Review (SEDAR) report, SEDAR 21. For the recreational fisheries, the final measures included a requirement for a shark endorsement for recreational permit holders, an online training requirement before obtaining the shark endorsement, additional education and outreach, and a requirement to use non-offset, non-stainless steel circle hooks while fishing for sharks within a specified geographic range unless using flies or artificial lures. Evidence suggests that circle hooks reduce at-vessel and post-release mortality rates for many HMS without reducing catch of target species compared to J-hooks. Circle hooks, by design, tend to hook sharks in the jaw more frequently than in the throat or gut (deep-hooking), thereby reducing injury and associated mortality compared to J-hooks (Willey et al. 2016; Godin et al. 2012, Campana et al. 2009). An outreach program to address bycatch and to educate anglers on the benefits of circle hooks has been implemented by NMFS. Several measures were included to educate anglers and reduce post-release mortality of dusky

sharks caught as bycatch by recreational fishermen. A video on the safe handling and release of prohibited Atlantic sharks is available at: <https://hmspermits.noaa.gov/sharkVideoEdu> and on the HMS permits website. Anglers and Charter-Headboat category permit holders must obtain a shark endorsement on their recreational permits in order to fish for, retain, possess or land sharks. Applicants must complete a brief online shark identification and fishing regulations training course and quiz prior to purchasing or renewing an applicable HMS Permit. In January 2011, NMFS created a brochure that provides guidelines on how to increase the survival of hook-and-line caught large pelagic species. This brochure was updated in 2017 as a result of finalization of Amendment 5b, and is available at: <https://www.fisheries.noaa.gov/resource/outreach-and-education/careful-catch-and-release-brochure>.

As of January 1, 2018, anglers fishing recreationally for sharks on a vessel with HMS Angling or HMS Charter-Headboat Permits must use non-offset, non-stainless steel circle hooks when fishing south of 41° 43' N latitude (near Chatham, Massachusetts, which is the northern extent of the dusky shark's U.S. Atlantic range), except when fishing with flies or artificial lures. Recreational anglers must also comply with other hook requirements. The 2006 Consolidated HMS FMP implemented a requirement effective January 1, 2007 that anglers fishing from an HMS-permitted vessel in any tournament awarding points or prizes for Atlantic billfish may deploy only non-offset circle hooks when using natural bait or natural bait/artificial lure combinations. The use of non-offset circle hooks increases the likelihood of post-release survival for billfish (Horodysky and Graves 2005) and reduces hook-related bleeding (Prince et al. 2002).

2.5 Commercial Fishing Permits

The type of permit(s) required to commercially harvest and sell HMS depends upon the species being targeted and the gear being used. A summary of the Atlantic HMS commercial permit requirements and the gear used by geographic area is summarized in the HMS Commercial Compliance Guide.

2.6 Recreational Fishing – Swordfish, Tunas, Billfish, and Sharks

Most Atlantic HMS are targeted by domestic recreational fishermen using a variety of handgear including rod and reel gear. To fish recreationally in federal waters for any Atlantic HMS, and within the waters of most Atlantic coastal states for Atlantic tunas, vessel owners must have a valid federal fishing permit for their vessel. The type of permit depends on the fish species, fishing gear, and fishing trip. The four types (or categories) of permits that can be used to recreationally fish for Atlantic HMS are HMS Angling, HMS Charter/Headboat, Atlantic tunas General category, and Swordfish General Commercial permit. All passengers on board a vessel with one of these valid HMS permits may recreationally fish for Atlantic HMS under applicable conditions. Only one of these four permits can be issued to a vessel in a calendar year, except that a vessel can be issued both an Atlantic tunas General category and Swordfish General Commercial permit in a calendar year. Permit holders may only change permit category within 10 days of the permit issuance date.

Federal recreational fishing regulations apply in federal waters and on the high seas, and may apply to recreational fishing in state waters. Anglers possessing a federal HMS fishing permit who are fishing in state waters must follow federal regulations for HMS, unless the state

regulations are more restrictive, in which case the state regulations apply. A summary of the Atlantic HMS recreational permit requirements and the gear used by geographic area is included in the HMS Recreational Compliance Guide found at <http://www.fisheries.noaa.gov/sfa/hms/compliance/guides/index.html>.

The recreational landings database for Atlantic HMS consists of information obtained through surveys including the Marine Recreational Information Program (MRIP), Large Pelagic Survey (LPS), Southeast Headboat Survey (HBS), Texas Headboat Survey, Recreational Billfish Survey (RBS) tournament data, and the HMS Recreational Reporting Program (non-tournament swordfish, billfish, and bluefin tuna). NMFS collects recreational catch-and-release data from dockside and telephone surveys (the LPS and MRIP) for the rod-and-reel fishery and uses these data to estimate total landings and discards. Statistical problems associated with small sample size remain an obstacle to estimating bycatch reliably in the rod-and-reel fishery. Coefficient of variations (CVs) can be high for many HMS (rare event species in the MRIP) and the LPS does not cover all times/geographic areas for non-bluefin tuna species. Unlike billfish, swordfish, or bluefin tuna, shark and BAYS tunas landings are not required to be reported to NMFS unless an angler is required to participate in LPS or MRIP. Descriptions of these surveys, the geographic areas they include, and their limitations are discussed in the 2006 Consolidated HMS FMP and previous HMS SAFE Reports.

2.7 Shark Research Fishery

As discussed above in Section 2.4.1, NMFS annually accepts applications to participate in the shark research fishery. From the applications received, NMFS randomly selects a small number of commercial vessels based upon certain criteria to participate in the shark research fishery. A valid shark research fishery permit is required to fish for, take, retain, or possess Atlantic sharks, including sandbar sharks, in excess of retention limits described in 50 CFR § 635.24(a). A shark research fishery permit is only valid for the vessel, owner, and operator(s) specified and cannot be transferred to another vessel, owner, or operator(s). A shark research fishery permit is only valid for the retention limits, time, area, and gear specified on the permit, and only when a NMFS-approved observer is on board. Although the shark research fishery is not restricted to applying to use only bottom longline gear, all participants to date have fished exclusively with bottom longline gear. The observer program for the shark research fishery was described in section 2.4.1. Issuance of a shark research fishery permit does not guarantee that the holder will be issued a NMFS-approved observer on any particular trip. Rather, issuance indicates that a vessel may be issued a NMFS-approved observer for a particular trip and on such trips may be allowed to harvest Atlantic sharks, including sandbar sharks, in excess of retention limits specified in § 635.24(a).

Except for the regulatory exemptions specifically referenced on the permit, all HMS regulations at 50 CFR Part 635 shall apply during the conduct of the fishing activity. All private vessels listed on a shark research permit should have a valid HMS recreational or commercial HMS permit. Fishermen with a shark research fishery permit should report their commercial catch in the appropriate logbook.

2.8 Fishing under Exempted Fishing Permits (EFPs), Scientific Research Permits (SRP), and Other Permits, and Associated Additional Gears Used

Regulations at 50 CFR § 600.745 and 50 CFR § 635.32 govern scientific research activity, exempted fishing, and exempted educational activity with respect to Atlantic HMS. EFPs, SRPs, and display permits are requested and issued under the authority of the MSA. NMFS issues EFPs, SRPs, and display permits to individuals conducting research or other fishing activities for HMS species using vessels that require exemptions from fishing regulations. For example, these permits may be necessary because possession of certain HMS species is restricted during many times of the year or because ICCAT requires reporting of all activities including scientific activities. Display permits are issued to individuals who are collecting HMS species for public display. 50 CFR 635.32(d). SRPs are required for scientific research activities concerning all species covered under 50 CFR part 635 regulated under the authority of the Atlantic Tunas Convention Act. 50 CFR 653.32(b). . Sometimes, the activities conducted under EFPs and SRPs is funded by NOAA to aid MSA management needs (e.g., Bycatch Reduction Engineering Program, Cooperative Research Program, Saltonstall-Kennedy Grant Program). Other times, the funding comes from private sources or from Universities that are conducting scientific research that will ultimately aid in NOAA stock assessments and management. When requested, NMFS provides Letters of Acknowledgement (LOAs) to those conducting scientific research activities from scientific research vessels (50 CFR 635.32(b); 50 CFR 600.745(a)); such activities are not subject to regulation under the MSA since they are not defined as “fishing” under 16 U.S.C. 1802(16). Letters of Acknowledgement do not authorize any activity, nor exempt it from regulations, but, rather, simply acknowledge it as scientific research. Thus, providing LOAs is not considered an agency action subject to Section 7 of the ESA and will not be considered further in this opinion.

While the majority of permits issued for research (e.g., EFPs and SRPs) use either commercial or recreational gear already authorized for these fisheries, a few use gear not otherwise generally authorized for Atlantic HMS. Mainly those gears include plankton nets and trawls and are used to collect either larvae or eggs. Bongo nets, neuston nets, and Multiple Opening/Closing Net and Environmental Sensing System (MOCNESS) are the typical plankton nets used to collect larval Atlantic HMS have limited sized openings and extremely small mesh. These nets are very selective and have very little unanticipated bycatch.

Another gear used by EFP applicants is the Methot frame trawl. The Methot frame trawl is a 5-m² aluminum frame with a 3.1-mm knotless mesh net. The nets used by most EFP applicants has a total length of 13.1-m (43 feet). The frame can be towed up to 5 knots. Floats may be attached to the bridle, as needed, to maintain a constant sampling depth. The net is deployed off the stern of the vessel and will be fished at a speed of approximately 4 knots. Typical tows last between 10-20 minutes, though that can be adjusted based on the size of catch. The net will be fished within 1-2 m of the surface and a flowmeter will be attached to estimate volume of water filtered. Bycatch associated with this trawl gear is very minimal.

Table 2.8 Gear Used for HMS EFPs, SRPs, and LOAs for HMS Issued 2016 (Permits listed multiple times if more than one gear type)

Gear type	Number of permits and letters
Pelagic longline	4
Bottom longline (including drumline)	23
Rod and reel and Handline	21
Purse seine	0
Plankton nets and trawl	2

Most EFPs, SRPs, and display permits involve fishing by commercial, recreational, or research vessels using fishing methods similar or identical to those used in the HMS fisheries. Under these circumstances, any effects from those activities would likely be similar to those analyzed in this Opinion. Each request includes a detailed description of the type of fishing and/or collection activities proposed, the gears to be used, and anticipated level of effort. If the fishing methods are similar, and the associated fishing effort does not represent a significant increase beyond the levels expected in the fishery described herein, then issuance of those EFPs, SRPs, and display permits would be expected to fall within the level of effort and impacts considered in this Opinion. For example, issuance of an EFP to an active commercial vessel is unlikely to add additional effects or increase fishing effort beyond what is otherwise likely to accrue from the vessel's normal commercial activities. Therefore, the issuance of EFPs, SRPs, and display permits for fishing consistent with the description of HMS fisheries analyzed in this Opinion is in most cases considered to be within the scope of this Opinion if it does not (1) increase fishing effort significantly or (2) have additional effects on listed species that are not considered in this Opinion. Directed research on any listed species (e.g., oceanic whitetip sharks) is not considered within the scope of this Opinion.

Each EFP, SRP, and display permit should be analyzed to determine whether the activity and effort fall within the scope of this Opinion. If so, any takes occurring during these activities would then be covered within the take anticipated in this opinion, and exempted from any take prohibition, within the parameters of the associated ITS. Applicants may be required to comply with terms and conditions or RPMs where relevant activities are being undertaken. The number of EFPs, and SRPs issued covering HMS from 2012 to 2018 by category are listed in Table 2.9.

Table 2.9 Number of Atlantic HMS EFPs and SRPs for HMS Issued 2012-2018

Permit type		2012	2013	2014	2015	2016	2017	2018
Exempted Fishing Permit	Sharks for display	4	4	3	3	3	5	6
	HMS** for display	2	2	3	1	0	2	2
	Tunas for display	0	0	0	0	0	0	0
	Shark research on a non-scientific vessel	10	10	10	11	12	4	4
	Tuna research on a non-scientific vessel	5	4	2	2	4	2	2
	HMS** research on a non-scientific vessel	3	3	3	4	4	4	2
	Billfish research on a non-scientific vessel	1	1	0	0	0	0	0
	Shark Fishing	0	0	0	0	0	0	0
	HMS** chartering	0	0	0	0	0	0	0
	Tuna fishing	0	0	1	1	0	0	0
	Total	25	24	22	22	23	17	16
Scientific Research Permit	Shark research	4	3	2	4	5	1	1
	Tuna research	3	2	2	1	1	0	1
	Billfish research	0	0	0	0	0	0	0
	HMS** research	4	3	3	1	1	3	6
	Total	11	8	7	6	7	4	8

*As of October 31, 2018.

**Multiple species

NMFS also issues permits for participation in the shark research fishery, discussed in the previous section, as EFPs. 50 CFR 635.32(f). Effort in the shark research fishery is evaluated in this Opinion in our analysis for shark bottom longline gear. In 2018, NMFS received 6 applications for the Shark Research Fishery permit. Based on the low number of applicants, NMFS issued EFPs to all 6 applicants.

2.9 Other Actions and Regulations Affecting the Proposed Action

2.9.1 Atlantic Large Whale Take Reduction Plan (ALWTRP)

Reducing large whale entanglement risks is the primary responsibility of the Atlantic Large Whale Take Reduction Team (ALWTRT). The ALWTRT was created in 1996 to address entanglement issues of large whales in fishing gear, including gill net gear. The ALWTRT was convened under the provisions of the MMPA, and through its efforts the ALWTRP was finalized in July 1997.

The ALWTRP is a plan promulgated under the MMPA to reduce serious injury and mortality (SI/M) to four large whale stocks that occur incidentally in certain fisheries. The target whale

stocks are the North Atlantic right whale western North Atlantic stock, humpback whale western North Atlantic stock, fin whale western North Atlantic stock, and minke whale Canadian East Coast stock.

To reduce serious injuries and mortality, the ALWTRP targets certain Category I and II fisheries under the MMPA's List of Fisheries (LOF). The LOF assigns specific categories to commercial fisheries based on their interactions with marine mammals. Category I designates fisheries with frequent serious injuries and mortalities incidental to commercial fishing; Category II designates fisheries with occasional serious injuries and mortalities incidental to commercial fishing; and Category III designates fisheries with a remote likelihood or no known serious injuries or mortalities incidental to commercial fishing.

The ALWTRP has several components, including restrictions on where and how gear can be set. It also requires research into whale populations and whale behavior, including research on fishing gear interactions and modifications that may lessen impacts to large whales. The ALWTRP also includes an outreach component to inform and collaborate with fishermen and a disentanglement program. The gillnet gear requirements under the ALWTRP differ for each management area and change based on location, season, and gear type depending on the species being protected. Portions of the ALWTRP specifically address the Atlantic shark fisheries. For more details or specific time/area gear regulations under the ALWTRP, please see 50 CFR § 229.32.

Major changes to the ALWTRP were implemented in a final rule that published on October 5, 2007 (72 FR 57104). Regulations that affect HMS fisheries, specifically gillnet fisheries, include: (1) a closed area for all gillnet fisheries from November 15 – April 15 from 29° 00' N to 32° 00' N from shore eastward to 80° 00' W and off SC, within 35 nmi of the coast (Southeast US Restricted Area North); (2) a restricted area from December 1 – March 31 from 27° 51' N to 29° 00' N from shore eastward to 80° 00' W (Southeast US Restricted Area South); (3) additional seasonal boundaries for EEZ waters east of 80° 00' W from 26° 46.50' N to 32° 00' N (Other Southeast Gillnet Waters); and (4) a monitoring area specific to the Atlantic shark gillnet fishery that extends from the area along the coast from 27° 51' N south to 26° 46.50' N eastward to 80° 00' W (Southeast US Monitoring Area) effective December 1 – March 31. Specific compliance requirements for fishing in these areas vary and are summarized in the Guide to the Atlantic Large Whale Take Reduction Plan (<https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-mammal-protection/atlantic-large-whale-take-reduction-plan#outreach>). The Plan has been modified on several occasions, most recently in 2015. For additional information, see the ALWTRP website <http://www.greateratlantic.fisheries.noaa.gov/Protected/whaletrp/> (NMFS 2017).

Amendment 9 to the 2006 Consolidated HMS FMP requires federal directed shark permit holders with gillnet gear on board to use VMS only in the Southeast U.S. Monitoring Area, pursuant to ALWTRP requirements. The Amendment 9 measures went into effect on March 15, 2016 (NMFS 2017).

2.9.2 Sea Turtle Handling and Resuscitation Techniques

NMFS published a final rule (66 FR 67495, December 31, 2001) detailing handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities. These techniques are meant to lessen the effects to sea turtles.

2.9.3 Mid-Atlantic Large-Mesh Gillnet Closure

NMFS published a final rule (67 FR 71895, December 3, 2002) enacting seasonal closures in the Mid-Atlantic EEZ for fishing with gillnets with a stretched mesh size of eight inches or greater, which was subsequently changed to seven inches or greater (71 FR 24776, April 26, 2006). The purpose of the action was to reduce the impact of large-mesh gillnet fisheries operating in areas where sea turtles were known to occur. Figure 2.6 shows the areas where the seasonal closures apply.

- Waters north of 33°51.0 N (North Carolina/South Carolina border at the coast) and south of 35°46.0 N (Oregon Inlet, North Carolina) at any time;
- Waters north of 35°46.0 N (Oregon Inlet, North Carolina) and south of 36°22.5 N (Currituck Beach Light, North Carolina) from March 16-January 14;
- Waters north of 36°22.5 N (Currituck Beach Light, North Carolina) and south of 37° 34.6 N (Wachapreague Inlet, Virginia) from April 1-January 14; and
- Waters north of 37° 34.6 N (Wachapreague Inlet, Virginia) and south of 37° 56.0 N (Chincoteague, Virginia) from April 16-January 14.

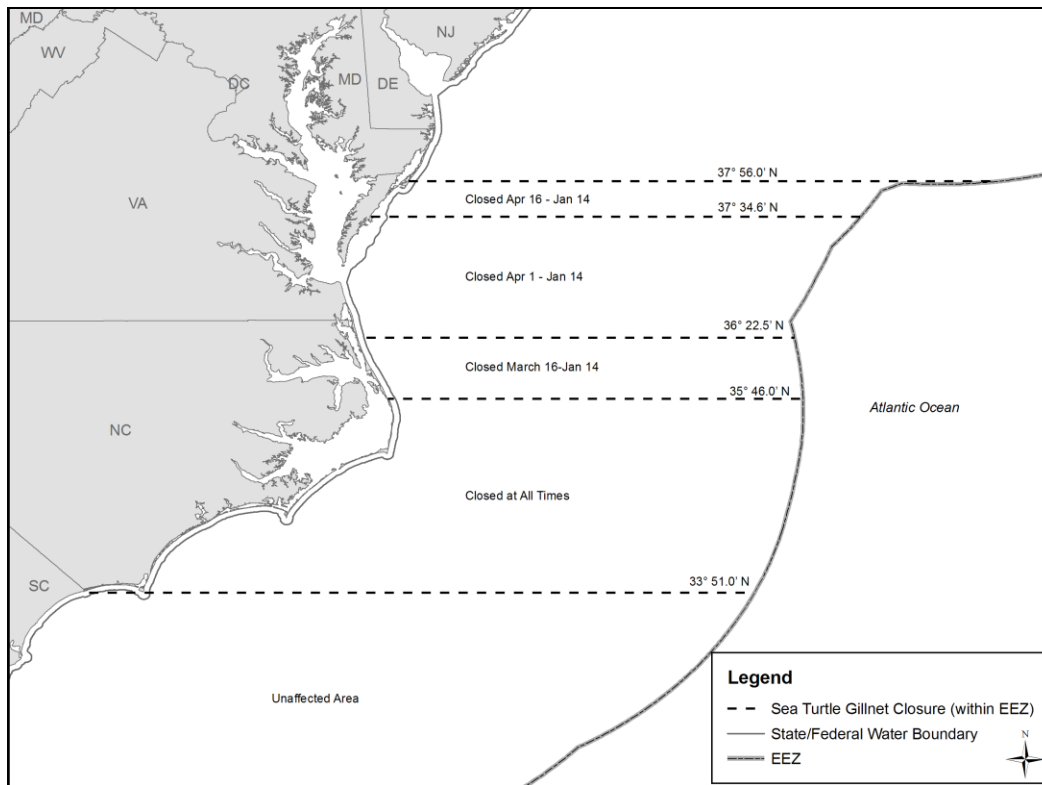


Figure 2.6 Mid-Atlantic Large Mesh Gillnet Closure Areas

2.10 Bycatch Mortality Reduction

The reduction of bycatch mortality is an important component of National Standard 9 of the MSA (16 USC 1851(a)(9)). National Standard 9 requires that fishery management plans minimize bycatch and, to the extent bycatch cannot be avoided, minimize the mortality of such bycatch. Atlantic HMS regulations require that all fish harvested from the management unit that are not retained must be released in a manner that will ensure maximum probability of survival, without removing the fish from the water. 50 CFR 635.21(a)(1). Research has shown that removing fish from the water significantly increases the likelihood of post-release mortality due to injuries associated with the stress of being hooked or caught in a net that are not immediately apparent. Because of these stress injuries, post-release mortality may not be anticipated by the fisherman who releases the fish, even in a rapid and safe manner. Ongoing research uses data on release techniques and from pop-up satellite tags to examine in situ mortality rates of Atlantic HMS. Information on bycatch mortality of these fish will continue to be collected and, in the future, may be used to estimate bycatch mortality in stock assessments. A summary of bycatch species, data collection methods, and management measures by fishery/gear type is found in Table 2.10. Additional details on bycatch management measures, observer coverage, bycatch and disposition, and protected species interactions in the HMS fisheries are reported in the HMS SAFE Report (NMFS 2017).

Table 2.10 Summary of Bycatch Species, MMPA List of Fisheries (LOF) Category, ESA Requirements, Data Collections, and Management Measures (Year Implemented) for the Atlantic HMS Fisheries

Fishery/Gear Type	Bycatch Species	MMPA LOF Category	ESA Requirements	Bycatch Data Collection	Bycatch-Related Management Measures
Shark bottom longline	Prohibited shark species, Target species after closure, Sea turtles, Smalltooth sawfish, Non-target finfish	Category III	ITS, Terms & Conditions, RPMs	Permit requirement (1993); logbook requirement (1993); observer coverage (1994)	Quotas (1993); trip limit (1994); gear marking (1999); handling & release guidelines (2001); line clippers, dipnets, corrodible hooks, de-hooking devices, move 1 nm after an interaction (2004); South Atlantic closure, VMS (2005); shark identification workshops for dealers (2007); sea turtle control device (2008); shark research fishery (2008); shark identification course for vessel owners and operators, move 1 nm after a dusky shark interaction and notify other vessels (2017); circle hooks (2018).
Northeast sink and Mid-Atlantic shark gillnet (smoothhound)	Marine mammals	Category I			Sink gillnet soak time limits and net check requirements for drift gillnets (2016)
Northeast, Southeast U.S. Atlantic, and Gulf of Mexico shark gillnet	Prohibited shark species, Sea turtles Marine mammals, Non-target finfish, Smalltooth sawfish	Category II	ITS, Terms & Conditions, RPMs	Permit requirement (1993); logbook requirement (1993); observer coverage (1994)	Quotas (1993); trip limit (1994); gear marking (1999); deployment restrictions (1999); 30-day closure for leatherbacks (2001); handling & release guidelines (2001); net checks, Southeast U.S. Restricted Area (2002); whale sighting (2002); VMS (2004; revised 2016); closure for right whale mortality (2006); shark identification workshops for dealers, Southeast U.S. Monitoring Area (2007); sink gillnet soak time limits and net check requirements for drift gillnets (2016); shark identification course for vessel owners and operators, move 1 nm after a dusky shark interaction and notify other vessels (2017).

Bluefin tuna purse seine	Undersize target species, Non-target finfish	Category III	ITS, Terms & Conditions	Permit requirement (1982); observer requirement (1996, 2001 only); EFPs (2002-03); VMS reporting (2015)	Quotas (1975); limited access, individual vessel quotas (1982); minimum size (1982); VMS trip declarations, bluefin retention and dead discard numbers and size(2015)
Bluefin tuna & swordfish harpoon	Undersize target species	Category III	ITS, Terms & Conditions	Permit requirement (bluefin tuna - 1982; swordfish - 1987); swordfish logbook requirement (1987); Online catch reporting (2015)	Quotas (bluefin tuna - 1982; swordfish- 1985); minimum size (bluefin - 1982; swordfish - 1985); Online catch reporting of bluefin retained and discarded dead (2015)
Handgear - commercial	Undersize target species, Non-target finfish	Category II	ITS, Terms & Conditions	Permit requirement (bluefin tuna - 1982; swordfish 1987; shark - 1993); logbook requirement (swordfish - 1985; shark - 1993); Online catch reporting (2015)	Regulations vary by species, including quotas, minimum sizes, retention limits, landing form; Online catch reporting of bluefin tuna discards and fish retained (2015).
Handgear – For-Hire	Undersize target species, Non-target finfish	Category III	ITS, Terms & Conditions	LPS (1992); MRFSS (1981); Online catch reporting (2015)	Regulations vary by species, including minimum sizes, retention limits, landing form; bluefin tuna quotas; Online catch reporting (2015); Circle hooks when fishing for sharks south of Chatham, MA; online shark identification and management measure video and quiz to obtain shark endorsement (2018).

MMPA – Marine Mammal Protection Act; ESA – Endangered Species Act; ITS – Incidental take statement; MRFSS – Marine Recreational Fishing Statistics Survey (now the Marine Recreational information Program or MRIP); EFPs – Exempted fishing permits; VMS – Vessel monitoring system; LPS – Large Pelagic Survey. NMFS 2017.

Source: NMFS 2017

2.10.1. Bluefin Tuna Purse Seine Fishery

NMFS has limited observer data on the bluefin tuna purse seine fishery due to inactivity in the fishery; however, when the fishery is active, data are collected through VMS, in which the vessel must declare the start and end of their trip and submit an HMS bluefin tuna catch report for each set, including the number of dead discards. There are no recorded instances of non-tuna finfish, other than minimal numbers of blue sharks, caught in tuna purse seines. Anecdotal evidence indicates that if fish are discarded, they are easily released out of the net with minimal bycatch mortality.

2.10.2. Shark Bottom Longline Fishery

The BLL fishery includes the shark research fishery, which is required to take an observer when targeting sandbar sharks, and the limited access fishery in which vessels are randomly selected for observer coverage and may be required to use a VMS. Vessel owners and operators must attend a protected species safe handling, release, and identification workshop every three years, must carry NMFS-approved dehooking devices onboard and use them in the event of a protected species interaction, and must store and post careful handling release protocols and guidelines in the wheelhouse to minimize injury to protected species when interactions occur. Any dusky shark or protected species that becomes entangled or hooked must be immediately released, and gear must be immediately retrieved and moved at least one nmi from that location before fishing is resumed to avoid interacting with the species again. Marine mammal entanglements must be reported to NMFS under the Marine Mammal Authorization Program. Time/area closures are implemented in this fishery to reduce bycatch, and require the proper stowage of gear if the vessel is within a closed area. BLL gear must use only corrodible hooks to prevent long-term injury of bycatch which cannot be released safely if the hook is removed. Disposition of discards and protected species interactions are recorded by observers and can be used to estimate discard mortality. Circle hooks were required starting in 2018. Observer coverage, bycatch and disposition, and protected species interactions in this fishery are reported in Section 5.5 of the HMS SAFE Report (NMFS 2017). NMFS collects data on the disposition (released alive or dead) of bycatch species from logbooks submitted by fishermen in the BLL fishery. Observer reports also include disposition of the catch as well as information on hook location, trailing gear, and injury status of protected species interactions.

2.10.3 Shark Gillnet Fisheries

Vessel owners and operators must attend a protected species safe handling, release, and identification workshop every three years. Fishermen using gillnet gear must limit soak times to 24 hours when using sink gillnet gear and conduct a net check at least every 2 hours when using drift gillnet gear to look for and remove any sea turtles, marine mammals, or smalltooth sawfish. If a marine mammal is taken, the vessel operator must immediately cease fishing operations and contact NMFS consistent with the Marine Mammal Authorization Program. Smalltooth sawfish must not be removed from the water while being removed from the net. Dusky sharks must be released immediately and vessels must move 1 nm after a dusky shark interaction and notify other vessels.

NMFS collects data on the disposition (released alive or dead) of bycatch species from logbooks submitted by fishermen in the shark gillnet fisheries. Observer reports include disposition of the catch, as well as information on injury status of protected species interactions, and can be used to estimate discard mortality.

2.10.4 HMS Commercial Handgear Fishery

Vessels targeting bluefin tuna with harpoon gear have not been selected for observer coverage since the deliberate fishing nature of the gear is such that bycatch is expected to be low. Bycatch in the swordfish harpoon fishery is expected to be virtually, if not totally, non-existent; therefore, bycatch mortality would be near zero. Disposition of bycatch reported in logbooks is used to estimate mortality of bycatch in the swordfish buoy gear fishery.

2.10.5 HMS Recreational Handgear Fishery

The LPS (dockside and telephone survey) collects data on disposition of bycatch (released alive or dead) in recreational Atlantic HMS fisheries from Virginia to Maine during June through October. Rod and reel discard estimates can be monitored through the expansion of survey data derived from the LPS, however, the actual numbers of fish discarded for many species are low. Post-release mortality estimation of billfishes has been examined in a review by Graves and Horodsky (2015). NMFS distributes educational outreach materials on the careful catch and release of Atlantic HMS to recreational fishing tournaments, where a large audience of recreational fishermen can be reached. To reduce dusky shark mortality, starting January 1, 2018, fishermen wishing to fish for sharks must watch an online shark identification video and take a quiz in order to obtain a shark endorsement on their Angling permit. These fishermen will also be required to use circle hooks when fishing for sharks south of Chatham, MA.

NMFS developed a Code of Angling Ethics as part of implementing Executive Order 12962 – Recreational Fisheries. NMFS implemented a national plan to support, develop, and implement programs that were designed to enhance public awareness and understanding of marine conservation issues relevant to the wellbeing of fishery resources in the context of marine recreational fishing. This code is consistent with National Standard 9, minimizing bycatch and bycatch mortality. These guidelines are discretionary, not mandatory, and are intended to inform the angling public of NMFS views regarding what constitutes ethical angling behavior. Part of the code covers catch-and-release fishing and is directed towards minimizing bycatch mortality. For a detailed description of the code, please refer to Section 3.9.8.3 of the 2006 Consolidated HMS FMP (NMFS 2006a).

2.11 Action Area

The action area for an Opinion is defined as the area affected by the federal action and not merely the immediate area involved in the action. Atlantic HMS fisheries are prosecuted under the Consolidated HMS FMP throughout the U.S. EEZ in the Atlantic Ocean, the Gulf of Mexico, and the Caribbean Sea. (Figures 2.1 and 2.2). Fishing areas are generally in the EEZ from the edge of the continental shelf and the shelf break and seaward (roughly 200 m and greater) and also influenced by the prevalence of major prevailing currents, confluences, upwelling zones and eddies.

3.0 Status of Listed Species and Critical Habitat

Table 3.1 Species and Critical Habitat that May Be Affected

Marine mammals	Scientific Name	Status
Blue whale	<i>Balaenoptera musculus</i>	Endangered
Fin whale	<i>Balaenoptera physalus</i>	Endangered
North Atlantic right whale	<i>Eubalaena glacialis</i>	Endangered
Sei whale	<i>Balaenoptera borealis</i>	Endangered
Sperm whale	<i>Physeter macrocephalus</i>	Endangered
Bryde's whale	<i>Balaenoptera edeni</i>	Endangered ^a
Sea Turtles	Scientific Name	Status
Green sea turtle	<i>Chelonia mydas</i>	Threatened ^b
Hawksbill sea turtle	<i>Eretmochelys imbricata</i>	Endangered
Leatherback sea turtle	<i>Dermochelys coriacea</i>	Endangered
Loggerhead sea turtle	<i>Caretta caretta</i>	Threatened ^c
Kemp's ridley sea turtle	<i>Lepidochelys kempii</i>	Threatened
Invertebrates		
Elkhorn coral	<i>Acropora palmata</i>	Threatened
Staghorn coral	<i>Acropora cervicornis</i>	Threatened
Rough cactus coral	<i>Mycetophyllia ferox</i>	Threatened
Pillar coral	<i>Dendrogyra cylindrus</i>	Threatened
Lobed star coral	<i>Orbicella annularis</i>	Threatened
Mountainous star coral	<i>Orbicella faveolata</i>	Threatened
Boulder star coral	<i>Orbicella franksi</i>	Threatened
Fish	Scientific Name	Status
Smalltooth sawfish	<i>Pristis pectinata</i>	Endangered ^d
Atlantic sturgeon	<i>Acipenser oxyrinchus oxyrinchus</i>	Endangered/Threatened ^e
Nassau grouper	<i>Epinephelus striatus</i>	Threatened
Scalloped hammerhead shark	<i>Sphyrna lewini</i>	Threatened ^f
Oceanic whitetip shark	<i>Cacharhinus longimanus</i>	Threatened
Giant manta ray	<i>Manta birostris</i>	Threatened
Shortnose sturgeon	<i>Acipenser brevirostrum</i>	Endangered
Gulf sturgeon	<i>Acipenser oxyrinchus desotoi</i>	Threatened
Gulf of Maine Atlantic Salmon	<i>Salmo salar</i>	Endangered
Critical Habitat		
Elkhorn and staghorn coral critical habitat		
Leatherback critical habitat		
Northwest Atlantic DPS of loggerhead sea turtle critical habitat		
^a Gulf of Mexico subspecies ^b The North Atlantic DPS and South Atlantic DPS ^c The Northwest Atlantic DPS ^d The U.S. DPS ^e The New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs are listed as endangered; the Gulf of Maine DPS is listed as threatened. ^f The Central Atlantic and Southwest Atlantic DPS		

3.1 Analysis of Species and Critical Habitat Not Likely to be Adversely Affected By the Proposed Action

We have determined that the proposed action being considered in this Opinion is not likely to adversely affect the following listed species or critical habitat: blue whales, sei whales, sperm whales, fin whales, Gulf of Mexico Bryde's whale, North Atlantic right whale, Gulf sturgeon, shortnose sturgeon, Gulf of Maine Atlantic salmon, Nassau grouper, elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, boulder coral, elkhorn and staghorn critical habitat, leatherback critical habitat, and NWA loggerhead DPS critical habitat. These species and critical habitats are therefore excluded from further analysis and consideration in this Opinion. The following discussion summarizes our rationale for these determinations.

3.1.1 Marine Mammals

Potential routes of effects to listed marine mammals from the proposed action include entanglement in fishing gear and collision with HMS fishing vessels, both of which could lead to injury or death. The degree of risk from fishing gear interactions is generally a function of the degree of spatial overlap between fishing effort and whale habitat, whale size and behavior, and the likelihood that an interaction will result in serious injury or mortality for a specific gear type (Benjamins et al. 2012). Vessel collisions with whales can occur where there is overlap between the vessel and the species. The risk of vessels strikes generally increases with increases in the number, size, and speed of vessels.

Fishing vessels actively fishing either operate at relatively slow speeds, drift, or remain idle, when setting, soaking, and hauling gear. Thus, any listed species in the path of a fishing vessel would likely have time to move away before being struck. Fishing vessels transiting to and from port or between fishing areas can travel at greater speeds, particularly recreational vessels, and thus do have more potential to strike a vulnerable species than during active fishing. However, given the rarity of listed marine mammal vessel strikes when considering (1) the large amount of overall vessel traffic in the action area, (2) that all fishing vessels represent only a portion of marine vessel activity and (3) that HMS fishing vessels represent an even smaller portion of marine activity, it seems extremely unlikely that a HMS vessel would strike a large whale, even during transiting. Based on this information, all listed marine mammals in the action area (blue, sei, sperm, fin whales, North Atlantic right whales, and Gulf of Mexico Bryde's whales) are not likely to be adversely affected by vessels fishing under the proposed action. Thus, for the remainder of 3.1.1, we only analyze potential effects from gear.

Blue, Sei, Sperm, and Fin Whales

The probability of blue, sei, sperm, and fin whales interacting with the proposed action is extremely low. Blue, sei, and sperm whales are predominantly found seaward of the continental shelf in deeper waters in the Atlantic and/or Gulf of Mexico and U.S. Caribbean (CETAP 1982; NMFS 2011c; Waring et al. 2013; Wenzel et al. 1988). Fin whales are generally found along the 100 m isobath with sightings also spread over deeper water including canyons along the shelf break (Waring et al. 2012). The gear types involved in the proposed action and the locations where they are fished make it extremely unlikely that these four whale species will interact with fisheries under this consultation. Gillnet and bottom longline gear are used outside of the

primary range or depth of these species. Gillnets targeting sharks in Southeast waters primarily operate in waters of approximately 9-21 m in depth, with an average depth of 13 m in the Gulf of Mexico from 2009-2013 (NMFS unpublished data). Gillnets targeting smoothhound sharks in the mid-Atlantic typically operate anywhere from state waters out approximately 20 miles (32 km) (Thorpe and Bereshoff 2000). Water depths at this distance from shore are approximately only 100 ft (30 m). This is outside the primary depth of these four whale species, which generally are found in deeper water. Shark bottom longline gear is typically fished in Southeast waters of approximately 15-62 m depths on average, with a reported average depth of 21 m in the Gulf of Mexico from 2007-2016 (NMFS unpublished data).

Other gears used in carrying out the proposed action, such as rod and reel, speargun, harpoon, green stick, buoy gear, bandit rigs, purse seine and handlines are unlikely to interact with the whales because of limited effort and/or gear setting techniques. These gears either consist of single lines set at specific depths for target species or are trolled (rod and reel, buoy gear, green stick, bandit rigs and handlines) or are sight fishing (speargun, harpoon, and purse seine) and can easily avoid whales, given their selectivity. Therefore, the gears used in the proposed action are not likely to adversely affect blue, sei, fin or sperm whales.

Gulf of Mexico Bryde's Whale

Gulf of Mexico Bryde's whales are extremely rare (estimated at fewer than 100 individuals), have a restricted distribution, and are the only resident baleen whale species in the Gulf of Mexico. The Gulf of Mexico Bryde's whale's range is a small area in the northeastern Gulf of Mexico near the De Soto Canyon (Rosel et al. 2016). The Bryde's whale Biologically Important Habitat Area (BIA) was identified in published literature as waters between 100 and 300 m depth along the continental shelf break (LaBrecque et al. 2015). However, given that there have also been sightings at 302 and 309 m depth in this region and west of Pensacola, Florida, the core area inhabited by the species is probably better described out to the 400 m depth contour and to Mobile Bay, Alabama, to provide some buffer around the deeper water sightings and to include all sighting locations in the northeastern Gulf of Mexico, respectively (Rosel et al., 2016). We consider this larger area, extending to the 400 m depth contour, an accurate description of the Gulf of Mexico Bryde's whale BIA, based on the recent sightings and tag data, and when we refer to the Gulf of Mexico Bryde's whale BIA, we are referring to this larger area.

Prior to listing the Gulf of Mexico Bryde's whale, NMFS reviewed the status of the species, including potential threats, and found that three commercial fisheries had the potential to interact with the Gulf of Mexico Bryde's whales given the gear types used and their general spatial distributions. The shark bottom longline fishery was identified as one such fishery. However, as described above, the majority of shark fishing (using both gillnets and bottom longlines) in the Gulf of Mexico occurs in waters shallower than 100 m, outside of the Bryde's whale BIA. More specifically, most shark bottom longline fishing effort occurs inshore of the Gulf of Mexico Bryde's whale habitat. For example, Soldevila et al. (2017) reported that throughout the eastern Gulf of Mexico, totals of 2498 and 3982 sets were observed by the shark fishery observer program during the periods 1994–2004 and 2005–2015, respectively, and of these, only 25 shark sets were observed within the BIA over 7 days during the 11-year period from 1994 to 2004. No observed sets occurred in the BIA from 2005 to 2015 (Soldevila et al. 2017). In addition, HMS fishery observers have not documented any Bryde's whale interactions or sightings with shark bottom longline gear. Thus given the depths and areas where HMS bottom longline gears are

fished, and the lack of observed interactions, they are extremely unlikely to affect the Gulf of Mexico Bryde's whales. Gillnet gear is not likely to adversely affect the species, given that lack of spatial overlap between the fishery and the species.

HMS fisherman targeting tunas and swordfish using other gear types (e.g., green stick, purse seines, harpoon, speargun, and vertical hook and line) are not concentrated in the Bryde's whale Habitat Area or northeastern Gulf of Mexico but rather are distributed throughout the action area as described in Section 2, and thus are not likely to adversely affect the species. In addition, we believe these gear types are not likely to adversely affect Gulf of Mexico Bryde's whale for the same reasons they are not likely to affect blue, sei, sperm, and fin whales; that is, these gears either consist of single lines set at specific depths for target species, are trolled (rod and reel, buoy gear, green stick, bandit rigs and handlines), or are sight fished (speargun, harpoon, and purse seine) and can easily avoid whales, given their selectivity. Due to the lack of recorded interactions or sightings during these HMS fishing activities, and the types of gear used and the depth where the proposed action occurs, we believe any effects on the Gulf of Mexico Bryde's whale from the proposed action are extremely unlikely.

North Atlantic right whale

HMS fishermen targeting tunas and swordfish using green stick, purse seines, harpoon, speargun, and vertical hook and line gears are distributed throughout the action area. However, we believe these gear types are not likely to adversely affect North Atlantic right whales for the same reasons described above for other listed whales.

Use of gillnet and bottom longline gears to target sharks has the potential to affect the North Atlantic right whale, however, interactions are extremely unlikely. The majority of the vessels and trips targeting sharks occur in the southern portion of the Atlantic region (which in Section 2.4.2 we refer to as the Southeast shark gillnet fishery). No large whale entanglements were documented or reported in Southeast shark gillnet gear from 2008-2017. Also no previous large whale entanglements can be definitively attributed to the smoothhound fishery (a gillnet fishery) operating in federal waters. The Southeastern U.S. shark gillnet fishery are listed as Category II fisheries in the 2018 List of Fisheries (83 FR 5389, February 7, 2018). Category II fisheries have been determined to have occasional incidental mortality and serious injury of marine mammals, causing annual mortality and serious injury greater than 1% and less than 50% of the PBR level for a given marine mammal stock. The shark bottom longline fishery is listed as a Category III fishery meaning it has an annual mortality and serious injury of a stock in a given fishery is less than or equal to 1 percent of the potential biological removal (PBR) level (i.e., a remote likelihood of or no known incidental mortality and serious injury of marine mammals). The only marine mammal interactions documented in the shark gillnet and bottom longline fisheries, upon which these classifications were based, however, were with bottlenose dolphins. Even considering these classifications, for the additional reasons below we believe interactions are extremely unlikely. In particular, as explained below, the measures put in place under the Atlantic Large Whale Take Reduction Plan (ALWTRP) make interactions with gillnets targeting shark species extremely unlikely.

ALWTRP

The ALWTRP currently recognizes seven gillnet areas: Cape Cod Bay Restricted Area, Great South Channel Restricted Gillnet Area, Great South Channel Sliver Restricted Gillnet Area,

Stellwagen Bank/Jeffreys Ledge Restricted Area, Other Northeast Gillnet Waters, Mid/South Atlantic Gillnet Waters, Southeast U.S. Restricted Area South, Southeast U.S. Monitoring Area, and Other Southeast Gillnet Waters.

Under the ALWTRP, certain restrictions apply to the South Atlantic gillnet fisheries; detailed regulations can be found at 50 CFR 229.32. No person may fish with or possess gillnet gear in the Southeast U.S. Restricted Area North during the restricted period (November 15 through April 15) (50 CFR 229.32(f)(2)(ii)). The Southeast U.S. Restricted Area North includes waters north of 29°00' N to 32°00' N (i.e., just south of Little River Inlet, South Carolina) and from the shoreline eastward to 80°00' W, and off the majority of South Carolina within 35 nmi of the shoreline. The only exemption for this area is for vessels transiting with gillnet gear aboard that have their nets covered with canvas or similar material; have their nets lashed or otherwise securely fastened to the deck, rail, or drum; have their buoys, high flyers, and anchors disconnected from all gillnets; and are in possession of no fish. Additionally, from December 1 through March 31, no person may fish with gillnet gear in the Southeast U.S. Restricted Area South (50 CFR 229.32(f)(2)(ii)(B)). The Southeast U.S. Restricted Area South includes waters north of 27°51' N. to 29°00' N and from the shoreline eastward to 80°00' W. Fishing with gillnet for sharks with webbing of 5 inches (12.7 cm) or greater stretched mesh is exempt from these restrictions from December 1-31 and from March 1-31, however, if the requirements found in 50 CFR 229.32(f)(2)(iii)(A)-(I) are met. Examples of regulations that are specific to shark gillnet fishing include: gillnet mesh size, requiring that drift gillnets remain attached to the vessel, the need to conduct net checks every two hours when drift gillnet gear is deployed, and a soak time limit of 24 hours for sink gillnets. The ALWTRP requires specific gear marking for southeastern gill nets.

The ALWTRP also includes management measures for the Mid-Atlantic gillnet fisheries. Per the ALWTRP, Mid/South Atlantic Gillnet Waters consists of all U.S. waters bounded on the north at 36°33.03' N from 72°30' W east to the eastern edge of the EEZ, and bounded on the south by 32°00' N east to the eastern edge of the EEZ (50 CFR 229.32(d)(7)). Regulations are as follows: from September 1 through May 31, no person may possess anchored gillnet gear unless that gear complies with the gear marking requirements specified in 50 CFR 229.32(d)(1) of the ALWTRP. Gear marking requirements for anchored gill nets (includes those weighted to the bottom of the sea) include: (1) no buoy line floating at the surface; (2) no wet storage of gear – anchored gear must be hauled out of the water at least once every 30 days; (3) gill net surface buoys must be marked to identify the vessel or fishery using at least 1 in height, block letters or Arabic numbers, in a color that contrasts with the color of the buoy; and (4) buoys must be marked with 1, 4-in blue mark midway along the buoy line. Additionally, all buoys, flotation devices, and/or weights must have a weak link having a maximum breaking strength of 1,100 lb, and all net panels are required to have a weak link with a maximum breaking strength of 1,100 lb in the center of the floatline of each 50-fathom net panel in a net string or every 25 fathoms for longer panels. Gillnets that do not return to port with the vessel must be anchored with the holding power of at least a 22-lb Danforth-style anchor at each end of the net string and must include weak link placement in 1 of 2 configuration options. Fishers are also encouraged to maintain their buoy lines to be as knot-free as possible. No drift gillnet gear may be fished at night unless gear is tended (i.e., attached to the vessel), and all drift gillnet gear must be removed from the water and stowed on board before returning to port (50 CFR 229.32 (e)(6)(ii)).

On January 22, 2006, a dead North Atlantic right whale calf was reported off Jacksonville, Florida. Based on the best available data, NMFS determined the whale's death had resulted from entanglement in allowable gillnet gear while inside the Southeast U.S. Restricted Area during the restricted period. In accordance with ALWTRP's implementing regulations at 50 CFR 229.32(g)(1), an emergency rule was issued on February 16, 2006, prohibiting all gillnet fishing within the Southeast U.S. Restricted Area (71 FR 8223). The prohibitions on gillnet fishing expired on March 31, 2006. Under the ALWTRP, closure of this area during North Atlantic right whale season (November 15 through March 31) must continue in perpetuity, unless other appropriate measures can be implemented to protect North Atlantic right whales.

In April of 2006, the Mid-Atlantic/Southeast Subgroup of the ALWTRT (SE Subgroup) was convened to discuss the North Atlantic right whale calf's death, the resultant emergency closure of the Southeast U.S. Restricted Area, and future management options that might avoid the total closure of this area in the future. The SE Subgroup suggested several potential management options that might allow the area to be reopened to gillnet fishing in the future.

Following these discussions, NMFS published a proposed rule on November 15, 2006 (71 FR 66485), amending the ALWTRP. Those proposed changes included expanding the Southeast U.S. Restricted Area to include waters within 35 nmi of the South Carolina coast; dividing the Southeast U.S. Restricted Area at 29°00' N into 2 areas— Southeast U.S. Restricted Areas North and South; and restricting gillnetting within the Southeast U.S. Restricted Area during the North Atlantic right whale calving season. Specifically, the rule proposed to prohibit gillnet fishing and possession in the Southeast U.S. Restricted Area North each year from November 15 through April 15, with an exemption for transiting through this area if gear is stowed in accordance with the rule. Additionally, gillnet fishing would be prohibited annually in the Southeast U.S. Restricted Area South from December 1 through March 31, with limited exemptions for gillnet fishing for sharks and Spanish mackerel.

Because the proposed protections would not be in place until well after North Atlantic right whales arrived in the Southeast U.S. Restricted Area for the 2006-2007 calving season, NMFS simultaneously published an emergency rule to protect North Atlantic right whales from entanglement in the core North Atlantic right whale calving area during right whale calving season (71 FR 66469, November 15, 2006). This emergency rule prohibited gillnet fishing or gillnet possession in Atlantic Ocean waters from the shore out to 80°00' W between 29°00' N and 32°00' N and within 35 nmi of the South Carolina coast. This emergency rule expired on April 15, 2007.

A rule published on June 25, 2007 (72 FR 34632), finalized the proposed amendments to the ALWTRP. The only difference between the proposed and Final Rules was an adjustment of the northern boundary of the Southeast U.S. Restricted Area to exclude Little River Inlet, South Carolina on the border between North Carolina and South Carolina (see Figure 5.1).

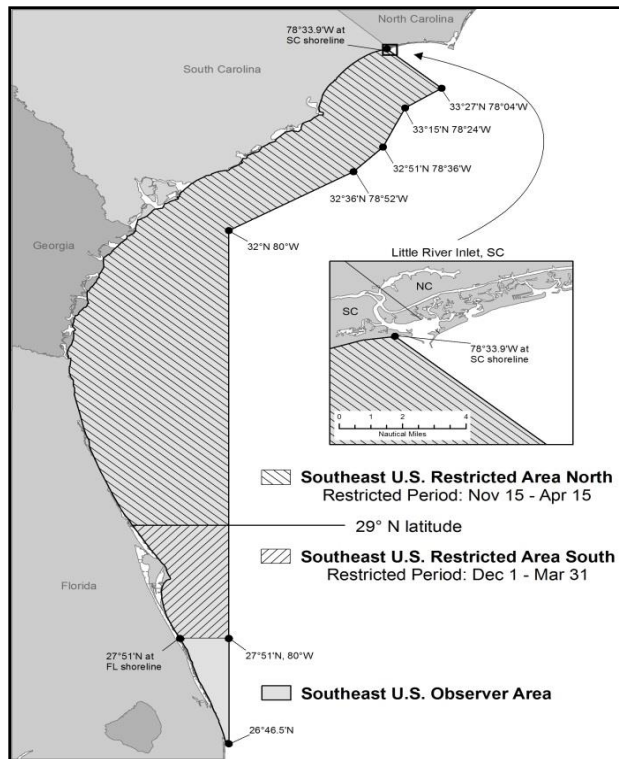


Figure 5.1. Southeast U.S. Restricted Area and restricted periods, as amended by the June 25, 2007 ALWTRP final rule (72 FR 34632)

NMFS believes these factors, in conjunction with known and predicted right whale distribution patterns in the Southeast U.S. Restricted Area south of 29° N. lat. during December through March, and existing Florida regulations prohibiting gillnetting in state waters that further reduce the potential spatial overlap between gillnet fishing and right whales, are operationally effective and will protect right whales from the risk of serious injury and mortality. Because the shark fisheries also underwent management changes in 2008 to prevent overfishing of Atlantic sharks (73 FR 40658, July 15, 2008), we believe data since 2008 is the best available data and timeframe to use for evaluating effects to North Atlantic right whales from the HMS shark gillnet fisheries in the Atlantic based on current management measures of both the 2006 Consolidated HMS FMP regulations and the ALWTRP regulations. As mentioned previously, gillnet effort targeting LCS and SCS declined as a result of Amendments 2 and 3 the Consolidated HMS FMP in 2007 and 2010. LCS and SCS targeted gillnet effort has continued to decline in the last five years (Carlson and Mathers 2017). With existing Atlantic shark gillnet practices, and continued management under the ALWTRP, we believe adverse effects on North Atlantic right whales from the Southeast shark gillnet fishery is extremely unlikely.

In addition to these southeast gillnet fisheries, in the northeast and mid-Atlantic regions, gillnet gear is the predominant gear type used in the smoothhound shark fishery, with smooth dogfish being primarily caught in the Mid-Atlantic region. Federal management of smoothhound sharks, which includes smooth dogfish and two other species in the smoothhound complex, was implemented through Amendment 9 to the 2006 Consolidated HMS FMP (November 24, 2015;

80 FR 46217) and began on March 15, 2016. Generally, fishermen use sink gillnet to target smooth dogfish in the northeast, although the species is often caught incidentally in bottom otter trawl gear as well (the latter of which is not subject to this consultation). The smooth dogfish sink gillnet fishery is a mixed fishery with a large portion of trips catching and retaining a variety of other species, dominated by bluefish, croaker, and spiny dogfish. Unlike the southeast and Gulf of Mexico regions, the northeast gillnet fisheries do not specifically target sharks in a given trip, but rather a variety of species in any given trip.

The best available information on the proportion of smoothhound landings in the sink gillnet fisheries, which was used in analyzing effects in this consultation, comes from Murray (2009b). Murray (2009b) reported that the adjusted average annual landings from all fisheries by sink gillnet gear in the Mid-Atlantic from 2002-2006 was 26,944 mt.⁶ Smoothhound landings from state and federal waters accounted for 5.89% (1,589 mt) of that total. Murray (2009b) included both state and federal information, and we used the distance from shore information provided by the VTR data to estimate smoothhound fishing effort in federal waters. From 2004-2011, the proportion of smoothhound landings coming from federal waters was the highest in 2004 at 47%, the lowest in 2007, 27%, and the mean was 36% (VTR Database, unpublished data).

NMFS GARFO analyzed take in all U.S. gillnet gear in their 2013 Batched Biological Opinion for Seven New England FMPs. Because there are mixed gillnet fisheries with the majority of them targeting a variety of species other than the HMS managed smoothhound, because smoothhound catch occurs primarily in state waters, and because of the restrictions in place on these fisheries described above, we believe it is highly unlikely that the smoothhound gillnets in federal waters would adversely affected North Atlantic right whales. In addition, attributing a portion of any of the gillnet interactions to the smoothhound fishery would result in overestimating interactions from the mixed gillnet fisheries that have been analyzed under the 2013 Opinion.

3.1.2 Gulf Sturgeon

Gulf sturgeon are not likely to be adversely affected by the proposed action. The Gulf sturgeon is an anadromous fish, inhabiting coastal rivers from Louisiana to Florida during the warmer months and over-wintering in estuaries, bays, and the Gulf of Mexico. Available data indicate Gulf sturgeon conduct alongshore migrations and primarily use shallow (2-6 m) nearshore areas as late wintering habitats (Edwards et al. 2007). They may also migrate out to the barrier islands; however, there is limited information on their migration habits. The fisheries in the proposed action operate offshore of these areas in deeper waters. Gulf sturgeon have never been observed caught in these fisheries. Based on this information, adverse effects from the proposed action are extremely unlikely to occur.

3.1.3 Shortnose Sturgeon

Shortnose sturgeon are benthic fish that mainly occupy the deep channel sections of large rivers and associated estuarine habitats, and unlike Atlantic and Gulf sturgeons, do not inhabit marine environments. They can be found in rivers along the western Atlantic coast from St. Johns

⁶ Estuarine anadromous fish breed in freshwater but otherwise live in estuarine environments.

River, Florida (possibly extirpated from this system), to the St. John River in New Brunswick, Canada. The species is estuarine anadromous⁷ in the southern portion of its range (i.e., south of Chesapeake Bay), while some northern populations are freshwater amphidromous⁸ (NMFS 1998). Since the fisheries in the proposed action do not operate in or near the rivers where concentrations of shortnose sturgeon are most likely found, it is highly unlikely that the fisheries will affect shortnose sturgeon.

3.1.4 Gulf of Maine Atlantic Salmon

The endangered Gulf of Maine Atlantic salmon DPS includes the wild population of Atlantic salmon of rivers and streams from the lower Kennebec River north to the U.S.-Canada border (i.e., Dennys, East Machias, Machias, Pleasant, Narraguagus, Ducktrap, and Sheepscot Rivers and Cove Brook). An anadromous species, juvenile salmon in New England rivers typically migrate to sea in May after a two- to three-year period of development in freshwater streams. The salmon remain at sea for two winters before returning to their U.S. natal rivers to spawn from mid-October through early November. While at sea, salmon generally undergo extensive migrations in the Northwest Atlantic to waters off Canada and Greenland, thus, they are widely distributed seasonally over much of the region. Although the 2006 Consolidated HMS FMP does authorize fishing within a portion of this species' range, there are no records of Atlantic salmon captures or other interactions in the HMS fishery observer, logbook, or survey data by any gear type. Captures of wild Atlantic salmon incidental to fishing for any species or by research/survey operations in the U.S. EEZ are exceedingly rare and, because there are no records of catch or other interactions in over twenty years of data reviewed for the HMS fisheries in these areas, the potential for the proposed action to affect Atlantic salmon via fishery interactions is extremely unlikely.

3.1.5 Nassau Grouper

The Nassau Grouper Biological Report (Hill and Sadovy de Mitcheson 2013) provides a detailed description of the species' distribution. The Nassau grouper's confirmed distribution currently includes "Bermuda and Florida (USA), throughout the Bahamas and Caribbean Sea" (Heemstra and Randall 1993). Nassau grouper is generally replaced ecologically in the eastern Gulf of Mexico by red grouper (*Epinephelus morio*) in areas north of Key West or the Tortugas (Smith 1971). They are considered a rare or transient species off Texas in the northwestern Gulf of Mexico (Gunter and Knapp 1951 in Hoese and Moore 1977). The first confirmed sighting of Nassau grouper in the Flower Garden Banks National Marine Sanctuary, which is located in the northwest Gulf of Mexico approximately 180 kilometers southeast of Galveston, Texas, was reported by Foley (2007). Many earlier reports of Nassau grouper up the Atlantic coast to North Carolina have not been confirmed.

The Nassau grouper is primarily a shallow-water, insular fish species that has long been valued as a major fishery resource throughout the wider Caribbean, South Florida, Bermuda, and the Bahamas (Carter et al. 1994). As larvae, the Nassau grouper is planktonic. After an average of

⁷ Estuarine anadromous fish breed in freshwater but otherwise live in estuarine environments.

⁸ Amphidromous fish make non-breeding movements between fresh and saltwater. Northern shortnose sturgeon do also ascend rivers for spawning.

35-40 days and at an average size of 32 mm TL, larvae recruit from an oceanic environment into demersal habitats (Colin 1992; Eggleston 1995).

Juvenile Nassau grouper (12-15 cm TL) are relatively solitary and remain in specific areas for months (Bardach 1958). Juveniles of this size class are associated with macroalgae, and both natural and artificial reef structures. As juveniles grow, they move progressively to deeper areas and offshore reefs (Colin et al. 1997; Tucker et al. 1993). Schools of 30-40 juveniles (25-35 cm TL) were observed at 8-10 m depths in the Cayman Islands (Tucker et al. 1993). No clear distinction can be made between types of adult and juvenile habitats, although a general size segregation with depth occurs; smaller Nassau grouper in shallower inshore waters (3.7-16.5 m) and larger individuals more common near deeper (18.3-54.9 m) offshore banks (Bardach 1958; Bardach et al. 1958; Cervigón 1994; Radakov et al. 1975; Silva Lee 1974; Thompson and Munro 1978).

Adult Nassau grouper tend to be relatively sedentary and are generally associated with high-relief coral reefs or rocky substrate in clear waters to depths of 130 m. Generally, adults are most common at depths less than 100 m (Hill and Sadovy de Mitcheson 2013) except when at spawning aggregations where they are known to descend to depths of 255 m (Starr et al. 2007). Adult Nassau grouper are unspecialized, bottom-dwelling, ambush-suction predators (Randall 1965; Thompson and Munro 1978).

Fishing gear targeting HMS is typically fished near the surface of the water or set to target pelagic species and avoid entanglement issues with bottom substrate, making it very unlikely that this gear will catch Nassau grouper or interact with benthic environments. Of the gear types used in the proposed action, only the shark bottom longline fishery occurs at deeper depths in the range of the Nassau grouper in the Florida Keys. There is limited fishing with the longline gear in this area compared to effort in other areas and there are no records of Nassau grouper in the fishery's observer program or logbook data. There are also no logbook or observer data catch records of Nassau grouper from other gear types used in the proposed action. Due to the method typically used to recreationally target HMS species near the surface in federal waters and the lack of Nassau grouper catch reported in these fisheries, we consider it unlikely that the HMS recreational fisheries would catch Nassau grouper. Because of the methods used to target HMS species primarily near the surface with most gear types and the lack of any attributable records of Nassau grouper catch to HMS gear, it is extremely unlikely that Nassau grouper will be adversely affected as a result of the proposed action.

3.1.6 Elkhorn, Staghorn, Rough Cactus, Pillar, Lobed Star, Mountainous Star, and Boulder Star Corals

We evaluated the potential effects of the proposed action on seven ESA-listed corals (Elkhorn, Staghorn, Rough Cactus, Pillar, Lobed Star, Mountainous Star, and Boulder Star Corals) based on the information provided in the species status reviews and the Final Listing Rules (71 FR 26852, May 09, 2006; 79 FR 53852, Sept. 10, 2014).

The known routes of effect from fishing on ESA-listed corals are a result of man-made abrasion and breakage resulting from vessel groundings, damaging fishing practices (and associated diver/snorkeler interactions and anchoring), and fishing/marine debris (ABRT et al. 2005). The

proposed action does not capture herbivorous fish, so there are no potential trophic effects to the listed corals.

Vessel groundings are possible as a result of the proposed action, but we believe these events are extremely unlikely to occur. Most of the commercial fishers participating in HMS fisheries are professional captains with years of experience operating vessels. Over the past 20 years, technological advancements and accessibility to depth gauges and GPS units have also increased vessel operators' ability to detect bottom features and calculate vessel position in relation to mapped coral structures. Experience and the use of technology greatly reduce the likelihood of vessels groundings. Additionally, some of these corals occur within the Florida Keys National Marine Sanctuary (FKNMS) (where prohibitions to injure or damage coral exist) or within 3 nmi of shore (i.e., and thus are not within the action area). FKNMS regulations govern the operations of vessels within its borders and prohibit vessels from striking or otherwise injuring corals (15 CFR 922.163(a)(5)(i)) (Table 3.2). The presence of navigational aids throughout the FKNMS is likely to further reduce the potential for vessel groundings. Given the experience of the vast majority of vessel operators, technology available, and the existence of navigational aids and regulations prohibiting vessel groundings, we believe adverse effects to and from such events are extremely unlikely to occur.

Within the area where these species and the proposed action overlap, only vertical line and spearguns are used or allowed. Thus, only the potential impacts from fishing operations utilizing these gear types under the proposed action are considered herein. The vertical line gear used in the proposed action is fished in water depths ranging from shallow estuaries to several hundred fathoms.

The information in Chiappone et al. (2005) suggests that the level of lost gear from hook-and-line fishing effort needed to impact coral is very high. They report that, while lost hook-and-line fishing gear was ubiquitous in the Florida Keys, it was estimated that < 0.2% of the milleporid hydrocorals, stony corals, and gorgonians in the habitats studied showed injury (e.g., colony abrasions and partial mortality) as a result of lost hook-and-line gear interactions. In Monroe County, Florida (i.e., the Florida Keys), the number of angler trips reporting landings of finfish (i.e., species likely to be targeting with hook-and-line gear) was 32,751 (<https://public.myfwc.com/FWRI/PFDM/ReportCreator.aspx>) for the year that Chiappone et al. (2005) conducted their study. This suggests that lost gear resulting from fishing effort of 32,751 sets per year, likely affected less than 0.2% of the milleporid hydrocorals, stony corals, and gorgonians.

Impacts to corals from hook-and-line fisheries interactions are most common to column and branching coral morphology that are more likely to become entangled by line or broken by gear. The rough cactus, lobed star, mountainous star, and boulder star coral species are characterized as boulder/mound or encrusting corals and area generally flat or round. In all cases, these species lack the branching morphology that greatly increases the potential risk of becoming fouled by fishing lines. We believe any adverse effects from fishing line entanglement associated with the proposed action to these four corals are extremely unlikely to occur. Pillar coral has protruding columns and the *Acropora* species have a branching morphology. However, given the low density of these listed corals where the fishing gear could occur, we expect the probability of interaction between HMS fishing gear and these species to be extremely low.

Spearguns are most commonly fished using SCUBA gear. Upon visually identifying a target fish, divers use rubber band guns or slings to hurl a spear shaft toward it. The HMS regulations implementing the 2006 Consolidated HMS FMP specify that speargun fishing gear may only be used when recreational fishing for Atlantic BAYS tunas and only from vessels issued either a valid HMS Angling or valid HMS Charter/Headboat permit. Persons fishing for Atlantic BAYS tunas using speargun gear, as specified at 50 CFR § 635.19, must be physically in the water when the speargun is fired or discharged, and may freedive, use SCUBA, or other underwater breathing devices. Only free-swimming BAYS tunas, not those restricted by fishing lines or other means, may be taken by speargun fishing gear. “Powerheads” or any other explosive devices, may not be used to harvest or fish for BAYS tunas with speargun fishing gear. 50 CFR 635.21(i).⁹

SCUBA divers (i.e., spearfishers) targeting HMS can accidentally damage corals. Also, speared fish may “hole up” under ledges, which may require spearfishers to come in close or direct contact with the bottom. However, impacts would generally be limited to a very temporary and extremely localized increase in sedimentation or incidental contact with the bottom. Those species of listed corals that are round/encrusting are less likely to be subject to significant damage by accidental contact or activity from divers. Spearfishers targeting HMS are generally competent divers, which further reduces the likelihood of accidental contact with all of the listed coral species (and greatly minimizes the potential for adverse effects) considered in this analysis. Additionally, in the FKNMS, there are regulations (Table 3.2) in place that prohibit damaging, breaking, cutting, or otherwise disturbing corals (15 CFR 922.163(a)(2)). FKNMS regulations also prohibit the taking or possessing of wildlife protected under the ESA (15 CFR 922.163(a)(10)). Mooring buoys have also been deployed throughout the FKNMS, reducing boaters’ need to anchor. Based on the general skill of the divers and the regulations in place to avoid and protect these corals, and the low probability of interaction with any of the species, we believe any adverse effects to listed coral species from spearfishers targeting HMS are extremely unlikely to occur. Regulations at 15 CFR 922.163 also prohibit the discharge of fishing/marine debris into the waters of the FKNMS. Regulations at 15 CFR 922.164 provide additional protection for corals occurring within existing management areas. Given the regulatory requirements, effects from spearfishing associated with the proposed action are considered extremely unlikely to occur.

⁹ Powerheads are underwater firearms that usually use 12-gauge or .357 Magnum rounds.

Table 3.2 Regulations Protecting Corals within the Florida Keys National Marine Sanctuary

Sanctuary Wide Prohibitions	
15 CFR § 922.163(a)(2)	<p><i>Removal of, injury to, or possession of coral or live rock.</i></p> <p>(i) Moving, removing, taking, harvesting, damaging, disturbing, breaking, cutting, or otherwise injuring, or possessing (regardless of where taken from) any living or dead coral, or coral formation, or attempting any of these activities, except as permitted under 50 CFR part 638.</p>
15 CFR § 922.163(a)(4)	<p><i>Discharge or deposit of materials or other matter.</i></p> <p>(i) Discharging or depositing, from within the boundary of the Sanctuary, any material or other matter, except:</p> <ul style="list-style-type: none"> (A) Fish, fish parts, chumming materials, or bait used or produced incidental to and while conducting a traditional fishing activity in the Sanctuary; (B) Biodegradable effluent incidental to vessel use and generated by a marine sanitation device approved in accordance with section 312 of the Federal Water Pollution Control Act, as amended, (FWPCA), 33 U.S.C. 1322 <i>et seq.</i>; (C) Water generated by routine vessel operations (e.g., deck wash down and graywater as defined in section 312 of the FWPCA), excluding oily wastes from bilge pumping; or (D) Cooling water from vessels or engine exhaust; <p>(ii) Discharging or depositing, from beyond the boundary of the Sanctuary, any material or other matter that subsequently enters the Sanctuary and injures a Sanctuary resource or quality, except those listed in paragraph (a)(4)(i) (A) through (D) of this section.</p>
15 CFR § 922.163(a)(5)	<p><i>Operation of vessels.</i></p> <p>(i) Operating a vessel in such a manner as to strike or otherwise injure coral, seagrass, or any other immobile organism attached to the seabed, including, but not limited to, operating a vessel in such a manner as to cause prop-scarring.</p> <p>(ii) Having a vessel anchored on living coral other than hardbottom in water depths less than 40 feet when visibility is such that the seabed can be seen.</p>

Table 3.2 Regulations Protecting Corals within the Florida Keys National Marine Sanctuary (continued)

Sanctuary Wide Prohibitions	
15 CFR § 922.163(a)(10)	<i>Take or possession of protected wildlife.</i> Taking any marine mammal, sea turtle, or seabird in or above the Sanctuary, <i>except</i> as authorized by the Marine Mammal Protection Act, as amended, (MMPA), 16 U.S.C. 1361 <i>et seq.</i> , the Endangered Species Act, as amended, (ESA), 16 U.S.C. 1531 <i>et seq.</i> , and the Migratory Bird Treaty Act, as amended, (MBTA) 16 U.S.C. 703 <i>et seq.</i>
Prohibitions Specific to Existing Management Areas	
15 CFR § 922.164(b)	<i>Key Largo and Looe Key Management Areas.</i> (i) Removing, taking, damaging, harmfully disturbing, breaking, cutting, spearing or similarly injuring any coral or other marine invertebrate, or any plant, soil, rock, or other material, except commercial taking of spiny lobster and stone crab by trap and recreational taking of spiny lobster by hand or by hand gear which is consistent with these regulations and the applicable regulations implementing the applicable Fishery Management Plan. (iii) Fishing with wire fish traps, bottom trawls, dredges, fish sleds, or similar vessel-towed or anchored bottom fishing gear or nets.
15 CFR § 922.164(d)(ii)	<i>Ecological Reserves and Sanctuary Preservation Areas.</i> Possessing, moving, harvesting, removing, taking, damaging, disturbing, breaking, cutting, spearing, or otherwise injuring any coral, marine invertebrate, fish, bottom formation, algae, seagrass or other living or dead organism, including shells, or attempting any of these activities.
15 CFR § 922.164(d)(v)	<i>Anchoring in the Tortugas Ecological Reserve.</i> In all other Ecological Reserves and Sanctuary Preservation Areas, placing any anchor in a way that allows the anchor or any portion of the anchor apparatus (including the anchor, chain or rope) to touch living or dead coral, or any attached living organism. When anchoring dive boats, the first diver down must inspect the anchor to ensure that it is not touching living or dead coral, and will not shift in such a way as to touch such coral or other attached organism. No further diving shall take place until the anchor is placed in accordance with these requirements.

To summarize, the unlikely interaction of the fisheries carried out as part of the proposed action with listed coral species, in combination with the measures in place to protect listed coral species where they do occur and avoid such interaction, makes any adverse effect on these species from the proposed action extremely unlikely to occur. Based on this information and the discussion provided in this section, effects on the listed coral species (Elkhorn, Staghorn, Rough Cactus,

Pillar, Lobed Star, Mountainous Star, and Boulder Star Corals) from the proposed action are extremely unlikely to occur.

3.1.7 Elkhorn and Staghorn Coral Critical Habitat

The physical or biological feature of elkhorn and staghorn coral critical habitat essential to their conservation is substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. Substrate of suitable quality and availability is defined as consolidated hardbottom or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover, occurring in water depths from the mean high water (MHW) line to 98 ft. 50 CFR 226.216.

Four areas of critical habitat were designated in Florida, Puerto Rico, St. Thomas/St. John, USVI, and St. Croix, USVI. The Florida area contains three sub-areas: (1) The shoreward boundary for Florida sub-area A begins at the 6-ft contour at the south side of Boynton Inlet, Palm Beach County at 26°32'42.5" N; then runs due east to the point of intersection with the 98-ft contour; then follows the 98-ft contour to the point of intersection with latitude 25°45'55" N, Government Cut, Miami-Dade County; then runs due west to the point of intersection with the 6-ft contour, then follows the 6-ft contour to the beginning point; (2) The shoreward boundary of Florida sub-area B begins at the MLW line at 25°45'55" N, Government Cut, Miami-Dade County; then runs due east to the point of intersection with the 98-ft contour; then follows the 98-ft contour to the point of intersection with longitude 82°W; then runs due north to the point of intersection with the South Atlantic Fishery Management Council (SAFMC) boundary at 24°31'35.75" N; then follows the SAFMC boundary to a point of intersection with the MLW line at Key West, Monroe County; then follows the MLW line, the SAFMC boundary (see 50 CFR 600.105(c)), and the COLREGS line (see 33 CFR 80.727, 730, 735, and 740) to the beginning point; and (3) The seaward boundary of Florida sub-area C (the Dry Tortugas) begins at the northern intersection of the 98-ft contour and longitude 82°45'W; then follows the 98-ft contour west around the Dry Tortugas, to the southern point of intersection with longitude 82°45'W; then runs due north to the beginning point. 50 CFR 226.216(b).

The Puerto Rico area includes all areas surrounding the islands of the Commonwealth of Puerto Rico, 98 ft in depth and shallower, seaward of the COLREGS line (see 33 CFR 80.738). The St. Thomas/St. John area and the St. Croix area includes all areas surrounding these islands, and smaller surrounding islands, where the water depths are 98 ft and shallower. 50 CFR 226.216(b).

Since floating gillnets (i.e., drift or strike nets) are fished near the surface and are not likely to come into contact with substrate of suitable quality and availability or dead coral skeleton, adverse effects from these gear types are extremely unlikely to occur. Recreational shark fishing targeting pelagic sharks troll hook-and-line gear at mid-water depths and are also extremely unlikely to come in contact with the essential feature of *Acropora* critical habitat. Bottom longlines and sink gillnets are fished at the bottom. However, we believe adverse effects to *Acropora* critical habitat essential features from these gears are extremely unlikely to occur. Bottom longline and sink gillnets are primarily used in sandy and muddy bottom habitats where the essential feature would not occur. Additionally, neither bottom longlines nor sink gillnets cause consolidated sediment to become unconsolidated, nor do they cause sedimentation or the

growth of macroalgae. For these reasons, we believe any adverse effects to designated critical habitat for elkhorn and staghorn are extremely unlikely to occur.

3.1.8 Leatherback Sea Turtle Critical Habitat

Critical habitat for leatherback sea turtles was designated to provide protection to sea turtles using the designated waters for courting and breeding, and as access to and from nesting areas on Sandy Point Beach, St. Croix, USVI. The area designated occurs in the waters adjacent to Sandy Point on the southwest corner of St. Croix, USVI, in waters from the 100-fathom curve shoreward to the level of mean high tide, with boundaries at 17°42'12"N and 64°50'00"W. (50 CFR 226.207). Due to the bathymetry around St. Croix, the water reaches the 100- fathom curve not far from the island's shore, thus, over 99% of the designated critical habitat adjacent to Sandy Point resides within USVI's waters. Thus, the proposed action has little to no overlap with the critical habitat area. Furthermore, as described in Chapter 2, HMS fisheries are limited in the Caribbean EEZ and most of the fishing in the Caribbean is recreational or artisanal vertical line ((handline, rod and reel). Given the small-scale of commercial HMS fisheries, the small size of the vessels involved, the relatively low number of known participants, and the use of traditional handgear, HMS fisheries in the Caribbean are considered to have negligible impacts to ocean and coastal habitats (NMFS 2012). Based on this information, we believe effects on leatherback critical habitat are extremely unlikely. Therefore, the proposed action is not likely to adversely affect leatherback sea turtle critical habitat.

3.1.9 Northwest Atlantic Loggerhead DPS Critical Habitat

Critical habitat for the NWADPS of loggerhead sea turtles in the South Atlantic is defined by 5 specific habitat types: nearshore reproductive, winter concentration, concentrated breeding, constricted migratory, and *Sargassum*. 50 CFR 226.223. Specifics of these habitats, including the primary constituent elements (PCEs) supporting each, can be found in Table 3.3.

The proposed action will have no effect on nearshore reproductive habitat (Units LOGG-N-3 through N-36) and winter concentration habitat (Units LOGG-N-1 and N-2). Nearshore reproductive habitats are those waters adjacent to nesting beaches and extend from the waterline out 1 mile. HMS fishers operate a minimum of 2 miles offshore of the 1-mile boundary, so there will be no possibility of impacting the PCEs of this critical habitat. Winter concentration habitat only occurs off the coast of North Carolina between Cape Hatteras and Cape Lookout. While HMS fishing occurs in this region, it is not capable of affecting the PCEs of water temperature, the proximity of shelf waters in relation to the Gulf Stream, and water depth.

NMFS designated two concentrated breeding habitat units (Units LOGG-N-17 and N-19) along the east coast of Florida as essential for the conservation of the species. The PCEs that support this habitat are (1) high densities of reproductive male and female loggerheads, (2) proximity to primary Florida migratory corridor, and (3) proximity to Florida nesting grounds.

The proposed action has the potential to capture protected loggerhead sea turtles in hook and line and gillnet gear as analyzed later in this Opinion, but we do not believe this will noticeably affect the density of reproductive males and females in the area. As discussed throughout section 5 of this Opinion, the proposed action may capture loggerhead sea turtles from various age classes across a large action area and approximately half of these sea turtles are expected to survive after

being released. Therefore, any effects on the first PCE are considered insignificant. Further, we believe the proposed action has no means by which to affect the other PCEs of concentrated breeding habitat. The gears and activities in these fisheries do not have the capacity to affect the distance of the concentrated breeding habitat in relation to the Florida migratory corridor or the Florida nesting grounds.

NMFS designated four constricted migratory habitat units along the east coast of Florida (Units LOGG-N-1 and LOGG-N-17 through N-19). The PCEs that support this critical habitat are (1) constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways, and (2) passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas.

The proposed action may operate within the constricted migratory corridor units. Given its activities and gear types, the proposed action does not have the capacity to modify the first PCE. The proposed action deploys gear in Atlantic waters that could possibly affect passage conditions (the second PCE). Yet, because any gears deployed in these areas fluctuate in time and space and are not permanent obstructions we do not expect them to meaningfully alter the passage conditions that allow migration to and from nesting, breeding, or feeding habitats. Any effects to the second PCE will be insignificant.

Two units of *Sargassum* critical habitat (LOGG-S-01 and LOGG-S-02) were designated to conserve loggerhead sea turtles by protecting essential forage, cover, and transport habitat for post-hatchlings and early juveniles. The PCEs that support this habitat are: (1) convergence zones, surface-water downwelling areas, the margins of major boundary currents, and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for the optimal growth of *Sargassum* and inhabitation of loggerheads, (2) *Sargassum* in concentrations that support adequate prey abundance and cover, (3) available prey and other material associated with *Sargassum* habitat, and (4) sufficient water depth and proximity to available currents to ensure offshore transport, foraging, and cover requirements for post-hatchlings.

The proposed action (all gear types and target species) could operate in the widespread areas of the *Sargassum* critical habitat units, but we believe any effects to the PCEs will be insignificant. The fishery does not have the capability to affect the location of convergence zones, surface-water downwelling (the movement of denser water downward in the water column) areas, or other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for optimal growth of *Sargassum* and inhabitation of loggerheads. The fishery would have no effect on availability of prey for hatchling loggerhead sea turtles or other material associated with *Sargassum* habitat because the fishery does not target or incidentally harvest smaller prey species or *Sargassum*. The fishery does not have the capability to affect the water depth or proximity to currents necessary for offshore transport, foraging, and cover. While some vessels associated with the proposed action may transit through *Sargassum* habitats, those vessel tracks are not anticipated to scatter *Sargassum* mats to the point of affecting the functionality of the PCEs. Further, the wakes and surface water disruption associated with these vessels are not of sufficient magnitude to result in significant effects to the distribution of *Sargassum* mats. Therefore, any adverse effects to the PCEs of *Sargassum* habitat would be insignificant.

In conclusion, activities associated with the proposed action are not likely to adversely affect any of the NWA loggerhead DPS critical habitat units. The proposed action will either have no effect on the critical habitat due to location or methods, or will have insignificant on the PCEs of critical habitat, and thus will not adversely affect the critical habitat.

Table 3.3. Details Regarding the PCEs of Critical Habitat for NWA DPS of Loggerhead Sea Turtles

Habitat Type	Units	State	Physical And Biological Features	Primary Constituent Elements
Nearshore Reproductive Habitat	LOGG-N-3, N-4, N-5, N-6	NC	Portion of nearshore waters adjacent to nesting beaches that hatchlings use as egress to the open-water environment. Also used by nesting females to transit between beach and open water during the nesting season.	1) Nearshore waters with direct proximity to nesting beaches that support critical aggregations of nesting turtles (e.g., highest density nesting beaches) to 1.6 kilometer (1 mile) offshore 2) Waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water 3) Waters with minimal manmade structures that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents
	LOGG-N-7, N-8, N-9, N-10, N-11	SC		
	LOGG-N-12, N-13	GA		
	LOGG-N-14, N-15, N-16, N-17, N-18, N-19, N-20, N-21, N-22, N-23, N-24, N-25, N-26, N-27, N-28, N-29, N-30, N-31, N-32	FL		
	LOGG-N-34, N-35, N-36	AL & MS		
Winter Concentration Habitat	LOGG-N-1, N-2	NC	Warm water habitat south of Cape Hatteras, near the western edge of the Gulf Stream, which supports meaningful aggregations of juveniles and adults during the winter months	1) Water temperatures above 10°C during the colder months of November through April 2) Continental shelf waters in proximity to the western boundary of the Gulf Stream 3) Water depths between 20-100 meters (m)
Concentrated Breeding Habitat	LOGG-N-17, N-19	FL	Sites that support meaningful aggregations of both male and female adult individuals during the breeding season	1) Meaningful concentrations of reproductive male and female loggerheads 2) Proximity to primary Florida migratory corridor 3) Proximity to Florida nesting grounds
Constricted Migratory Corridor Habitat	LOGG-N-1	NC	High-use migratory corridors that are constricted (limited in width) by land on 1 side and the edge of the continental shelf and Gulf Stream on the other side	1) Constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways 2) Passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas
	LOGG-N-17, N-18, N-19	FL		
<i>Sargassum</i> Habitat	LOGG-S-1, S-2	Atlantic Ocean & Gulf of Mexico	Developmental and foraging habitat for young loggerheads where surface waters form accumulations of floating material, especially <i>Sargassum</i>	1) Convergence zones, surface-water downwelling areas, and other locations where there are concentrated components of the <i>Sargassum</i> community in water temperatures suitable for optimal growth of <i>Sargassum</i> and inhabitance of loggerheads 2) <i>Sargassum</i> in concentrations that support adequate prey abundance and cover 3) Available prey and other material associated with <i>Sargassum</i> habitat such as, but not limited to, plants and cyanobacteria and animals endemic to the <i>Sargassum</i> community such as hydroids and copepods 4) Sufficient water depth and proximity to available currents to ensure offshore transport, and foraging and cover requirements by <i>Sargassum</i> for post-hatchling loggerheads (i.e., >10 m depth to ensure not in surf zone)

3.2 Analysis of HMS Gear Types that are Not Likely to Adversely Affect Some or all ESA-Listed Species

Although sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead shark, oceanic whitetip sharks, and giant manta rays are all likely to be adversely affected by at least some gear types, or vessels, associated with the proposed action evaluated in this Opinion, certain HMS gear types, or vessels, are not likely to adversely affect all or some of these species. Table 3.4 provides an overview of the gear types associated with the proposed action and whether they are likely to adversely affect or not likely to adversely affect sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays.

Table 3.4. HMS Gear Types that are Not Likely to Adversely Affect (NLAA) or Likely to Adversely Affect (LAA) ESA-listed Species.

Gear Type	Sea Turtles	Smalltooth Sawfish	Atlantic Sturgeon	Scalloped Hammerhead Central and Southwest Atlantic DPS	Oceanic Whitetip Shark	Giant Manta Ray
Purse Seine	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Speargun	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Harpoon	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Green-stick	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Rod and Reel	LAA	LAA	NLAA	LAA	LAA	NLAA
Bandit	LAA	NLAA	NLAA	LAA	LAA	NLAA
Buoy Gear	LAA	NLAA	NLAA	LAA	LAA	NLAA
Handline	LAA	NLAA	NLAA	LAA	LAA	NLAA
Bottom Longline	LAA	LAA	NLAA	NLAA	NLAA	LAA
Gillnet	LAA	LAA	LAA	NLAA	NLAA	LAA
Vessels	LAA	NLAA	NLAA	NLAA	NLAA	NLAA

The following sections provide our analysis of the HMS gear types that are not likely to adversely affect ESA-listed species. Section 5 of this Opinion analyzes the HMS gear types and vessel activities that are likely to adversely affect ESA-listed species.

3.2.1 Purse seine gear

There are no records of interactions between the bluefin tuna purse seine gear and any listed species in the action area. Generally, purse seine gear is a pelagic gear used to target species such as herring, mackerel, and tuna. Similar to midwater trawl gear, purse seine gear has a negligible catch of multispecies, as the gear is designed to fish in the upper layers of the water column for fish schooling at or near the surface of the ocean. In addition, as opposed to trawl gear, purse seine gear is not towed through the water column, giving demersal species (such as sawfish) the opportunity to escape the gear.

The bluefin tuna purse seine fishery is listed as a category III fishery under the MMPA. Purse seines are set when a school of fish is located, then the vessel pays out the net in a circle around the school. This affords considerable control over what is encircled by the net and the net does not remain set in the water for an appreciable amount of time.

Five vessels are authorized to fish for bluefin tuna as authorized under Purse Seine category permits. The fishery has been largely inactive in recent years, with no landings since 2015. When active, the gear type is very selective in targeting bluefin tuna, has very limited effort, and observers have not reported capture of species other than bluefin tuna while prosecuting the fishery. Listed species have never been captured. For these reasons, even if the fishery were active, we believe that future entanglements, like current entanglements, are extremely unlikely to occur. Use of this gear associated with the proposed action is not likely to adversely affect any listed species described in Table 3.4 (i.e., sea turtles, smalltooth sawfish, Atlantic sturgeon, Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray).

3.2.2 Spearguns

Under the HMS regulations, speargun gear is authorized to recreationally target BAYS tunas. Persons fishing for Atlantic BAYS tunas using speargun gear must be physically in the water when the speargun is fired or discharged, and may freedive, use SCUBA, or other underwater breathing devices. BAYS tunas must be free-swimming and cannot be restricted by fishing lines or other means. “Powerheads” may not be used. No other HMS may be taken with speargun fishing gear, including bluefin tuna, swordfish, sharks, sailfish, spearfish, roundscale spearfish, or white or blue marlin (50 CFR 635.21(i)).

Divers spearfish by visually detecting and shooting BAYS tunas at close proximity. The maximum operational range of a spear is about 9-13 ft (about 3 to 4 m)—less than that if the spear is fitted with a powerhead. It is highly unlikely that divers would be within 13 ft of certain listed species analyzed in this Opinion, such as a sea turtle, or that they would accidentally shoot a listed species while in such close proximity. On extremely rare occasions, divers may encounter listed species at a moderate- to long-distance while diving. In these instances, there may be potential behavioral effects to the species (e.g., change in swim speed or direction, curious approaches); however, these effects are expected to be temporary and insignificant. For all of these reasons, we believe listed species noted in Table 3.4 (i.e., sea turtles, smalltooth sawfish, Atlantic sturgeon, Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray) are not likely to be adversely affected from permitted speargun gear fishing associated with the proposed action.

3.2.3 Harpoon

There are no historic or recent reports of interactions with sea turtles, smalltooth sawfish, Atlantic sturgeon, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetips shark, or giant manta rays while fishing for HMS with permitted harpoon gear. Harpoon gear includes a pointed dart or iron attached to the end of a line several hundred feet in length, the other end of which is attached to a floatation device. Harpoon gear is attached to a pole that is propelled only by hand and not by mechanical

means. Harpoon gear is used to target bluefin tuna, BAYS tunas, and swordfish from vessels at close range. Fishermen would be able to identify target species before deployment and therefore it is extremely unlikely that harpoon gear would interact with the ESA-listed in Table 3.4. For all of these reasons, we believe use of harpoon gear to target HMS associated with the proposed actions is not likely to adversely affect listed species noted in Table 3.4.

3.2.4 Green-Stick Gear

There are no historic or recent reports of interactions with listed species by vessels fishing HMS with green stick gear. As described in Section 2, green-stick gear is authorized under the proposed action for use targeting bluefin tuna, BAYS tunas, and swordfish. This gear includes an actively trolled mainline attached to a vessel and elevated or suspended above the surface of the water by a “green stick” with no more than 10 hooks or gangions attached to the mainline. The suspended line, attached gangions and/or hooks, and catch may be retrieved collectively by hand or mechanical means. All of the species in Table 3.4 are not likely to be adversely affect by this action, including sharks and sea turtles that are known to be captured on baited hooks. Since green-stick gear is actively trolled and often uses artificial lures where the fishermen are tending the gears that the boat is towing through the water (these are not the typical bait of turtles or sharks), it is extremely unlikely that this gear type would interact with any of the ESA-listed species described in Table 3.4.

3.2.5 Vertical Line Gear - Rod and Reel, Bandit Gear, Buoy Gear, Handline Gear

Vertical hook and line, including rod and reel, bandit gear, buoy gear, and handline gear, used to target HMS both commercially and recreationally in the EEZ is not likely to adversely affect Atlantic sturgeon or giant manta rays. All of these gears except rod and reel are not likely to adversely affect smalltooth sawfish. Commercial handgears are often used to fish for Atlantic HMS by fishermen on private vessels, charter vessels, and headboat vessels. With the exception of rod and reel gear, these gears tend to be very selective and are deployed while schools of the target species are actively feeding around the fishing vessel. They also are not fished on the ocean bottom. Due to the selectivity of these gears, and the way they are fished, we believe effects from bandit gear, buoy gear, and handline gear are extremely unlikely to interact with, and thus are not likely to adversely affect, Atlantic sturgeon, giant manta ray, and smalltooth sawfish.

Rod and reel gear used to target HMS in the EEZ is not likely to adversely affect Atlantic sturgeon. Rod and reel gear may be deployed from a vessel that is at anchor, drifting, or underway (i.e., trolling). Most directed HMS fishing effort in the action area takes place by trolling or drifting for pelagic species. In general, trolling consists of dragging baits or lures through, on top of, or even above the water’s surface. Atlantic sturgeon are generally considered to be benthic species and not likely to be in the water column where anglers targeting pelagic species would set their gear. Additionally, recreational fishermen targeting HMS do not use Atlantic sturgeon prey species as bait. The benthic nature of this species and its feeding habits and prey preferences lead us to believe adverse effects from this gear are extremely unlikely to occur.

Vertical hook-and-line gear (rod and reel, bandit, buoy, and handline) used to target HMS in the EEZ is not likely to adversely affect giant manta rays. HMS fishing is not occurring in areas of known aggregations of giant manta rays in the action area. Additionally, giant manta rays are primarily filter feeders and not expected to be attracted to baits used. Adverse effects from HMS vertical line gear on giant manta rays are extremely unlikely to occur.

The use of vertical line gears is likely to adversely affect sea turtles, oceanic whitetip sharks, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, and smalltooth sawfish (rod and reel only), as discussed in Section 5.

3.2.6 Bottom Longline Gear

The use of bottom longline gear associated with the proposed action is not likely to adversely affect Atlantic sturgeon, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, or oceanic whitetip sharks. Sea turtles, smalltooth sawfish and giant manta rays are known to be adversely affected by bottom longline gear, and those effects are discussed in Section 5 of this document.

Because of their diet and feeding mechanism, Atlantic sturgeon are not likely to feed on baited hooks. Atlantic sturgeon are described generally as eating both plants and animals off the surface of the water bottom. In the marine environment, Atlantic sturgeon feed on mollusks, polychaete worms, gastropods, shrimps, amphipods, isopods, and small fish (Scott and Crossman 1973a). These species are not used as bait in the shark bottom longline fishery meaning Atlantic sturgeon are unlikely to be attracted to the gear, minimizing the likelihood of hooking and/or entanglement. For these reasons, adverse effects from this gear are extremely unlikely to occur.

There are no records of the Central and Southwest Atlantic DPS of scalloped hammerhead shark or oceanic whitetip shark being caught by HMS bottom longline gear. Observer and logbook records indicate that bottom longline gear is not currently used by HMS permit holders outside of the U.S. EEZ surrounding the continental United States (or within the U.S. EEZ in Caribbean). From 2008-2013, there was no reported use of bottom longline gear by HMS permit holders in the U.S. Caribbean. Several year-round time and area closures in the Caribbean are in effect for this gear, which likely limit use of this gear. 50 CFR 635.21(d)(1)(ii). Data indicate that bottom longline gear is not used within the range of the Central and Southwest Atlantic DPS of scalloped hammerhead shark. The gear may be used within the range of oceanic whitetip sharks, however, oceanic whitetip sharks are more commonly found towards the surface of the water column and there are no records of oceanic whitetip sharks being caught in shark bottom longline gear from the observer program. For these reasons, we believe adverse effects from this gear on the Central and Southwest Atlantic DPS of scalloped hammerhead shark and oceanic whitetip are extremely unlikely to occur.

3.2.7 Gillnets

The use of gillnet gear is not likely to adversely affect the Central and Southwest Atlantic DPS of scalloped hammerhead shark or oceanic whitetip sharks. Sea turtles, smalltooth sawfish, Atlantic sturgeon, and giant manta rays are likely to be adversely affected by this gear, and those effects are discussed in Section 5 of this document.

There was no reported use of gillnets in the U.S. EEZ in the Caribbean between 2008 and 2015. There are several year-round closures for gillnet gear in specific areas of the U.S. EEZ in the Caribbean (50 CFR 622.435(b)(2)), which likely limit use of this gear. Since the Central and Southwest Atlantic DPS of scalloped hammerhead shark is predominantly found in areas where gillnet use has not been recorded or is limited, the potential for interaction with this gear is extremely low. In addition, there are no records of the Central and Southwest Atlantic DPS of scalloped hammerhead shark or oceanic whitetip sharks being caught by HMS gillnet gear. Oceanic whitetip sharks generally occur farther offshore than gillnet usage and are found at the top of the water column (gillnets are fished at the bottom) and therefore do not overlap with areas of gillnet effort. For these reasons, we believe adverse effects to the Central and Southwest Atlantic DPS of scalloped hammerhead shark and oceanic whitetip sharks from this gear are extremely unlikely to occur.

3.2.8 Vessels

Smalltooth sawfish and Atlantic sturgeon spend the vast majority of their time at or near the seafloor, where they are not vulnerable and subject to vessel interactions. Their benthic habits make it extremely unlikely that these species would be struck by or otherwise interact with a vessel.

The Central and Southwest Atlantic DPS of scalloped hammerhead sharks, oceanic whitetips sharks, and giant manta rays spend time in the water column and do not need to surface to breathe, making it unlikely that they would be struck by or otherwise subject to vessel interactions. Although giant manta rays have shown evidence of vessel interactions in near shore aggregation areas where giant manta rays and vessels may be concentrated, we do not believe that vessels associated with the proposed action that are transiting to and fishing in the EEZ are likely to strike a giant manta ray because the vessels associated with the proposed action do not transit through giant manta ray aggregation areas. Even if scalloped hammerhead sharks, oceanic whitetip sharks, or giant manta rays are found at the surface, they are highly mobile species and likely able to avoid a vessel strike. Thus, the effects of the fishing vessels used in the commercial and recreational HMS fisheries analyzed in this Opinion, in terms of species interactions or strikes by the vessels themselves, are not likely to adversely affect the Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip, and giant manta ray.

As discussed in Section 5 of this document, sea turtles are likely to be adversely affected by vessels associated with the proposed action.

3.3 Analysis of Species Likely to be Adversely Affected

3.3.1 General Threats Faced by All Sea Turtle Species

Sea turtles face numerous natural and man-made threats that shape their status and affect their ability to recover. Many of the threats are either the same or similar in nature for all listed sea turtle species, those identified in this section are discussed in a general sense for all sea turtles. Threat information specific to a particular species are then discussed in the corresponding status sections where appropriate.

Fisheries

Incidental bycatch in commercial fisheries is identified as a major contributor to past declines, and threat to future recovery, for all of the ESA-listed sea turtle species in the action area (NMFS and USFWS 1991; NMFS and USFWS 1992; NMFS and USFWS 1993; NMFS and USFWS 2008; NMFS et al. 2011). Domestic fisheries often capture, injure, and kill sea turtles at various life stages. Sea turtles in the pelagic environment are exposed to U.S. Atlantic pelagic longline fisheries. Sea turtles in the benthic environment in waters off the coastal United States are exposed to a suite of other fisheries in federal and state waters. These fishing methods include trawls, gillnets, purse seines, hook-and-line gear (including bottom longlines and vertical lines [e.g., bandit gear, handlines, and rod-reel]), pound nets, and trap fisheries. Refer to the Environmental Baseline section of this opinion for more specific information regarding federal and state managed fisheries affecting sea turtles within the action area). The Southeast U.S. shrimp fisheries have historically been the largest fishery threat to benthic sea turtles in the southeastern United States, and continue to interact with and kill large numbers of sea turtles each year.

In addition to domestic fisheries, sea turtles are subject to direct as well as incidental capture in numerous foreign fisheries, further impeding the ability of sea turtles to survive and recover on a global scale. For example, pelagic stage sea turtles, especially loggerheads and leatherbacks, circumnavigating the Atlantic are susceptible to international longline fisheries including the Azorean, Spanish, and various other fleets (Aguilar et al. 1994; Bolten et al. 1994). Bottom longlines and gillnet fishing is known to occur in many foreign waters, including (but not limited to) the northwest Atlantic, western Mediterranean, South America, West Africa, Central America, and the Caribbean. Shrimp trawl fisheries are also occurring off the shores of numerous foreign countries and pose a significant threat to sea turtles similar to the impacts seen in U.S. waters. Many unreported takes or incomplete records by foreign fleets make it difficult to characterize the total impact that international fishing pressure is having on listed sea turtles. Nevertheless, international fisheries represent a continuing threat to sea turtle survival and recovery throughout their respective ranges.

Non-Fishery In-Water Activities

There are also many non-fishery impacts affecting the status of sea turtle species, both in the ocean and on land. In nearshore waters of the United States, the construction and maintenance of federal navigation channels has been identified as a source of sea turtle mortality. Hopper dredges, which are frequently used in ocean bar channels and sometimes in harbor channels and offshore borrow areas, move relatively rapidly and can entrain and kill sea turtles (NMFS 1997). Sea turtles entering coastal or inshore areas have

also been affected by entrainment in the cooling-water systems of electrical generating plants. Other nearshore threats include harassment and/or injury resulting from private and commercial vessel operations, military detonations and training exercises, in-water construction activities, and scientific research activities.

Coastal Development and Erosion Control

Coastal development can deter or interfere with nesting, affect nesting success, and degrade nesting habitats for sea turtles. Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). These factors may decrease the amount of nesting area available to females and change the natural behaviors of both adults and hatchlings, directly or indirectly, through loss of beach habitat or changing thermal profiles and increasing erosion, respectively (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). In addition, coastal development is usually accompanied by artificial lighting which can alter the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings that are drawn away from the water (Witherington and Bjorndal 1991). In-water erosion control structures such as breakwaters, groins, and jetties can impact nesting females and hatchlings as they approach and leave the surf zone or head out to sea by creating physical blockage, concentrating predators, creating longshore currents, and disrupting of wave patterns.

Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., dichlorodiphenyltrichloroethane [DDT], polychlorinated biphenyls [PCB], and perfluorinated chemicals [PFC]), and others that may cause adverse health effects to sea turtles (Garrett 2004; Grant and Ross 2002; Hartwell 2004; Iwata et al. 1993). Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges may directly injure individuals through skin contact with oils (Geraci 1990), inhalation at the water's surface and ingesting compounds while feeding (Matkin and Saulitis 1997). Hydrocarbons also have the potential to impact prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area.

The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (DWH Trustees 2015). Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. The spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. Information on the spill impacts to individual sea turtle species is presented in the Status of the Species sections for each species.

Marine debris is a continuing problem for sea turtles. Sea turtles living in the pelagic environment commonly eat or become entangled in marine debris (e.g., tar balls, plastic bags/pellets, balloons, and ghost fishing gear) as they feed along oceanographic fronts where debris and their natural food items converge. This is especially problematic for sea turtles that spend all or significant portions of their life cycle in the pelagic environment (i.e., leatherbacks, juvenile loggerheads, and juvenile green turtles).

Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change, exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see <http://www.climate.gov>).

Climate change impacts on sea turtles currently cannot be predicted with any degree of certainty; however, significant impacts to the hatchling sex ratios of sea turtles may result (NMFS and USFWS 2007a). In sea turtles, sex is determined by the ambient sand temperature (during the middle third of incubation) with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25°-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007a).

The effects from increased temperatures may be intensified on developed nesting beaches where shoreline armoring and construction have denuded vegetation. Erosion control structures could potentially result in the permanent loss of nesting beach habitat or deter nesting females (NRC 1990). These impacts will be exacerbated by sea level rise. If females nest on the seaward side of the erosion control structures, nests may be exposed to repeated tidal overwash (NMFS and USFWS 2007b). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005). 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).

Other changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish, etc.) which could ultimately affect the primary foraging areas of sea turtles.

Other Threats

Predation by various land predators is a threat to developing nests and emerging hatchlings. The major natural predators of sea turtle nests are mammals, including raccoons, dogs,

pigs, skunks, and badgers. Emergent hatchlings are preyed upon by these mammals as well as ghost crabs, laughing gulls, and the exotic South American fire ant (*Solenopsis invicta*). In addition to natural predation, direct harvest of eggs and adults from beaches in foreign countries continues to be a problem for various sea turtle species throughout their ranges (NMFS and USFWS 2008).

Diseases, toxic blooms from algae and other microorganisms, and cold stunning events are additional sources of mortality that can range from local and limited to wide-scale and impacting hundreds or thousands of animals.

3.3.2 Loggerhead Sea Turtles – Northwest Atlantic DPS

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. NMFS and USFWS published a Final Rule which designated 9 DPSs for loggerhead sea turtles (76 FR 58868, September 22, 2011, and effective October 24, 2011). This rule listed the following DPSs: (1) Northwest Atlantic Ocean (threatened), (2) Northeast Atlantic Ocean (endangered), (3) South Atlantic Ocean (threatened), (4) Mediterranean Sea (endangered), (5) North Pacific Ocean (endangered), (6) South Pacific Ocean (endangered), (7) North Indian Ocean (endangered), (8) Southeast Indo-Pacific Ocean (endangered), and (9) Southwest Indian Ocean (threatened). The Northwest Atlantic (NWA) DPS is the only one that occurs within the action area, and therefore it is the only one considered in this Opinion.

Species Description and Distribution

Loggerheads are large sea turtles. Adults in the southeast U.S. average about 3 ft (92 cm) long, measured as a straight carapace length (SCL), and weigh approximately 255 lb (116 kg) (Ehrhart and Yoder 1978). Adult and subadult loggerhead sea turtles typically have a light yellow plastron and a reddish brown carapace covered by non-overlapping scutes that meet along seam lines. They typically have 11 or 12 pairs of marginal scutes, 5 pairs of costals, 5 vertebrales, and a nuchal (precentral) scute that is in contact with the first pair of costal scutes (Dodd Jr. 1988).

The loggerhead sea turtle inhabits continental shelf and estuarine environments throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd Jr. 1988). Habitat uses within these areas vary by life stage. Juveniles are omnivorous and forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd Jr. 1988). Subadult and adult loggerheads are primarily found in coastal waters and eat benthic invertebrates such as mollusks and decapod crustaceans in hard bottom habitats.

The majority of loggerhead nesting occurs at the western rims of the Atlantic and Indian Oceans concentrated in the north and south temperate zones and subtropics (NRC 1990). For the NWA DPS, most nesting occurs along the coast of the U.S., from southern Virginia to Alabama. Additional nesting beaches for this DPS are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba

(Moncada Gavilan 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

Non-nesting, adult female loggerheads are reported throughout the U.S. Atlantic, Gulf of Mexico, and Caribbean Sea. Little is known about the distribution of adult males who are seasonally abundant near nesting beaches. Aerial surveys suggest that loggerheads as a whole are distributed in U.S. waters as follows: 54% off the southeast U.S. coast, 29% off the northeast U.S. coast, 12% in the eastern Gulf of Mexico, and 5% in the western Gulf of Mexico (TEWG 1998a).

Within the NWA DPS, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf Coast of Florida. Previous Section 7 analyses have recognized at least 5 western Atlantic subpopulations, divided geographically as follows: (1) a Northern nesting subpopulation, occurring from North Carolina to northeast Florida at about 29°N; (2) a South Florida nesting subpopulation, occurring from 29°N on the east coast of the state to Sarasota on the west coast; (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez M. 1990; TEWG 2000a); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS 2001b).

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles concluded that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida Peninsula. It also concluded that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the recovery plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are as follows: (1) the Northern Recovery Unit (Florida/Georgia border north through southern Virginia), (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The recovery plan concluded that all recovery units are essential to the recovery of the species. Although the recovery plan was written prior to the listing of the NWA DPS, the recovery units for what was then termed the Northwest Atlantic population apply to the NWA DPS.

Life History Information

The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, which include the ecosystems those stages generally use: (1) egg (terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone¹⁰), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting

¹⁰ Neritic refers to the nearshore marine environment from the surface to the sea floor where water depths do not exceed 200 meters.

female (terrestrial zone) (NMFS and USFWS 2008). Loggerheads are long-lived animals. They reach sexual maturity between 20-38 years of age, although age of maturity varies widely among populations (Frazer and Ehrhart 1985; NMFS 2001b). The annual mating season occurs from late March to early June, and female turtles lay eggs throughout the summer months. Females deposit an average of 4.1 nests within a nesting season (Murphy and Hopkins 1984), but an individual female only nests every 3.7 years on average (Tucker 2010). Each nest contains an average of 100-126 eggs (Dodd Jr. 1988) which incubate for 42-75 days before hatching (NMFS and USFWS 2008). Loggerhead hatchlings are 1.5-2 inches long and weigh about 0.7 oz. (20 g).

As post-hatchlings, loggerheads hatched on U.S. beaches enter the “oceanic juvenile” life stage, migrating offshore and becoming associated with *Sargassum* habitats, driftlines, and other convergence zones (Carr 1986; Conant et al. 2009; Witherington 2002). Oceanic juveniles grow at rates of 1-2 inches (2.9-5.4 cm) per year (Bjorndal et al. 2003; Snover 2002) over a period as long as 7-12 years (Bolten et al. 1998) before moving to more coastal habitats. Studies have suggested that not all loggerhead sea turtles follow the model of circumnavigating the North Atlantic Gyre as pelagic juveniles, followed by permanent settlement into benthic environments (Bolten and Witherington 2003; Laurent et al. 1998). These studies suggest some turtles may either remain in the oceanic habitat in the North Atlantic longer than hypothesized, or they move back and forth between oceanic and coastal habitats interchangeably (Witzell 2002). Stranding records indicate that when immature loggerheads reach 15-24 in (40-60 cm) SCL, they begin to reside in coastal inshore waters of the continental shelf throughout the U.S. Atlantic and Gulf of Mexico (Witzell 2002).

After departing the oceanic zone, neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Estuarine waters of the U.S., including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, Florida Bay, as well as numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads (Conant et al. 2009).

Like juveniles, non-nesting adult loggerheads also use the neritic zone. However, these adult loggerheads do not use the relatively enclosed shallow-water estuarine habitats with limited ocean access as frequently as juveniles. Areas such as Pamlico Sound, North Carolina, and the Indian River Lagoon, Florida, are regularly used by juveniles but not by adult loggerheads. Adult loggerheads do tend to use estuarine areas with more open ocean access, such as the Chesapeake Bay in the U.S. Mid-Atlantic. Shallow-water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads (Conant et al. 2009).

Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of Mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months,

and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has also been documented (Hawkes et al. 2007; Georgia Department of Natural Resources, unpublished data; South Carolina Department of Natural Resources, unpublished data). Satellite telemetry has identified the shelf waters along the west Florida coast, The Bahamas, Cuba, and the Yucatán Peninsula as important resident areas for adult female loggerheads that nest in Florida (Foley et al. 2008a; Girard et al. 2009; Hart et al. 2012). The southern edge of the Grand Bahama Bank is important habitat for loggerheads nesting on the Cay Sal Bank in The Bahamas, but nesting females are also resident in the bights of Eleuthera, Long Island, and Ragged Islands. They also reside in Florida Bay in the U.S., and along the north coast of Cuba (A. Bolten and K. Bjorndal, University of Florida, unpublished data). Moncada et al. (2010) report the recapture of 5 adult female loggerheads in Cuban waters originally flipper-tagged in Quintana Roo, Mexico, which indicates that Cuban shelf waters likely also provide foraging habitat for adult females that nest in Mexico.

Status and Population Dynamics

A number of stock assessments and similar reviews (Conant et al. 2009; Heppell et al. 2003a; NMFS-SEFSC 2009; NMFS 2001b; NMFS and USFWS 2008; TEWG 1998a; TEWG 2000a; TEWG 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

Numbers of nests and nesting females can vary widely from year to year. Nesting beach surveys, though, can provide a reliable assessment of trends in the adult female population, due to the strong nest site fidelity of female loggerhead sea turtles, as long as such studies are sufficiently long and survey effort and methods are standardized (e.g., (NMFS and USFWS 2008). NMFS and USFWS (2008) concluded that the lack of change in 2 important demographic parameters of loggerheads, remigration interval and clutch frequency, indicate that time series on numbers of nests can provide reliable information on trends in the female population.

Peninsular Florida Recovery Unit

The Peninsular Florida Recovery Unit (PFRU) is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed an average of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (NMFS and USFWS 2008). The statewide estimated total for 2017 was 96,912 nests (FWRI nesting database).

In addition to the total nest count estimates, the Florida Fish and Wildlife Research Institute (FWRI) uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. This provides a better tool for understanding the nesting trends (Figure 3.2). FWRI performed a detailed analysis of the long-term loggerhead index nesting data (1989-2017; <http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trend/>). Over that time period, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that

was followed by a sharp decline over the subsequent 9 years. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represents a new record for loggerheads on the core index beaches. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but nonsignificant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening confidence intervals (<http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trend/>). Nesting at the core index beaches declined in 2017 to 48,033, and rose slightly again to 48,983 in 2018, which is still the 4th highest total since 2001.

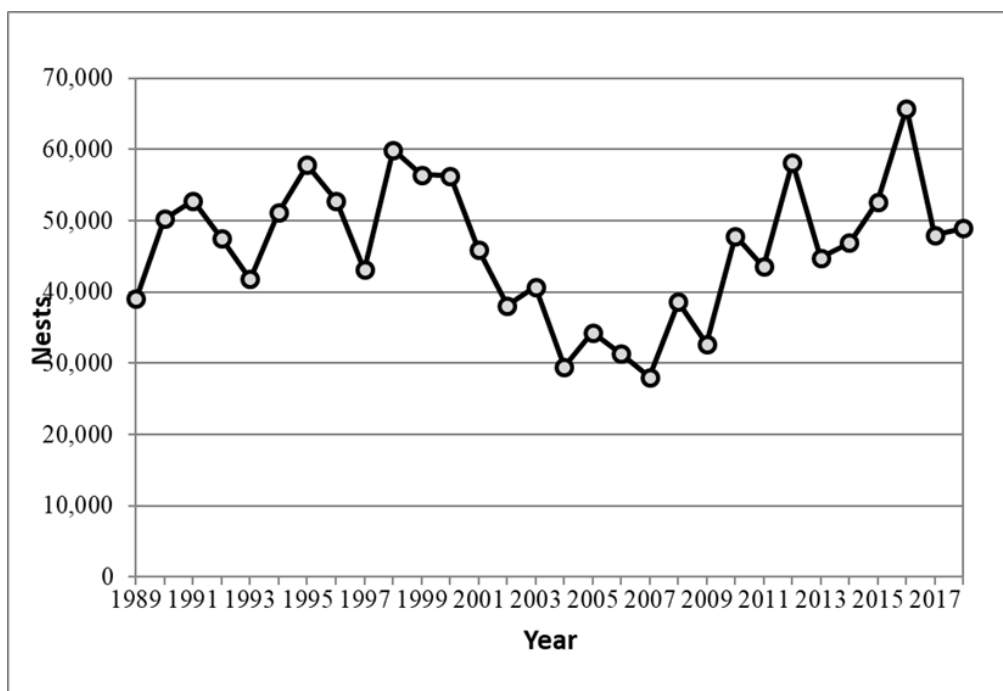


Figure 3.2 Loggerhead sea turtle nesting at Florida index beaches since 1989

Northern Recovery Unit

Annual nest totals from beaches within the Northern Recovery Unit (NRU) averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches (Georgia Department of Natural Resources [GADNR] unpublished data, North Carolina Wildlife Resources Commission [NCWRC] unpublished data, South Carolina Department of Natural Resources [SCDNR] unpublished data), and represent approximately 1,272 nesting females per year, assuming 4.1 nests per female (Murphy and Hopkins 1984). The loggerhead nesting trend from daily beach surveys showed a significant decline of 1.3% annually from 1989-2008. Nest totals from aerial surveys conducted by SCDNR showed a 1.9% annual decline in nesting in South Carolina from 1980-2008. Overall, there are strong statistical data to suggest the NRU had experienced a long-term decline over that period of time.

Data since that analysis (Table 3.5) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, GADNR press release, <http://www.georgiawildlife.com/node/3139>). South Carolina and North Carolina nesting have also begun to shift away from the past declining trend. Loggerhead nesting in Georgia broke a record 2013 and South Carolina and North Carolina broke records in 2015. Nesting in all three states then topped those records again in 2016. Nesting in 2017 and 2018 declined relative to 2016, but preliminary 2019 estimates indicate 2019 will break new records in all three states again (Klemm, per comm., December 10, 2019).

Table 3.5 Total Number of NRU Loggerhead Nests Recorded (GADNR, SCDNR, and NCWRC nesting datasets compiled at Seaturtle.org)

Nests	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
GA	1,649	998	1,760	1,992	2,241	2,289	1,196	2,319	3,265	2,155	1,735
SC	4,500	2,182	3,141	4,015	4,615	5,193	2,083	5,104	6,443	5,232	2,762
NC	841	302	856	950	1,074	1,260	542	1,254	1,612	1,195	765
Total	6,990	3,472	5,757	6,957	7,930	8,742	3,821	8,677	11,320	8,582	5,262

South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record

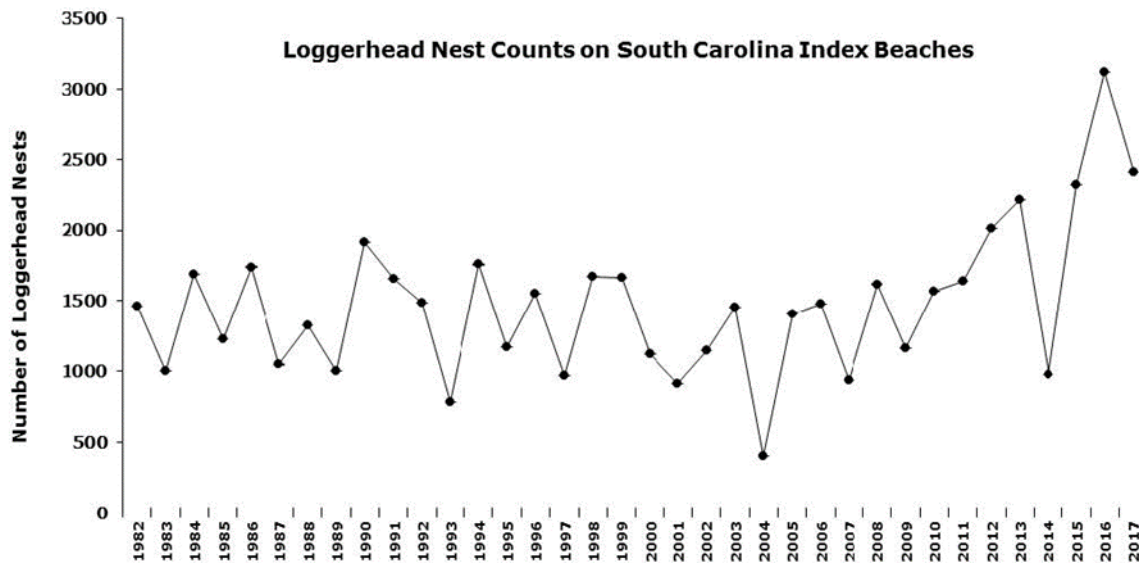


Figure 3.4 South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: <http://www.dnr.sc.gov/seaturtle/nest.htm>)

Other Northwest Atlantic DPS Recovery Units

The remaining 3 recovery units—Dry Tortugas (DTRU), Northern Gulf of Mexico (NGMRU), and Greater Caribbean (GCRU)—are much smaller nesting assemblages, but they are still considered essential to the continued existence of the species. Nesting surveys for the DTRU are conducted as part of Florida’s statewide survey program. Survey effort was relatively stable during the 9-year period from 1995-2004, although the 2002 year was missed. Nest counts ranged from 168-270, with a mean of 246, but there was no detectable trend during this period (NMFS and USFWS 2008). Nest counts for the NGMRU are focused on index beaches rather than all beaches where nesting occurs. Analysis of the 12-year dataset (1997-2008) of index nesting beaches in the area shows a statistically significant declining trend of 4.7% annually. Nesting on the Florida Panhandle index beaches, which represents the majority of NGMRU nesting, had shown a large increase in 2008, but then declined again in 2009 and 2010 before rising back to a level similar to the 2003-2007 average in 2011. Nesting survey effort has been inconsistent among the GCRU nesting beaches, and no trend can be determined for this subpopulation (NMFS and USFWS 2008). Zurita et al. (2003) found a statistically significant increase in the number of nests on 7 of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. Nonetheless, nesting has declined since 2001, and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008).

In-water Trends

Nesting data are the best current indicator of sea turtle population trends, but in-water data also provide some insight. In-water research suggests the abundance of neritic juvenile loggerheads is steady or increasing. Although Ehrhart et al. (2007) found no significant regression-line trend in a long-term dataset, researchers have observed notable increases in CPUE (Arendt et al. 2009; Ehrhart et al. 2007; Epperly et al. 2007). Researchers believe

that this increase in CPUE is likely linked to an increase in juvenile abundance, although it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. Bjorndal et al. (2005), cited in NMFS and USFWS (2008), caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern U.S. may be due to increased abundance of the largest oceanic/neritic juveniles (historically referred to as small benthic juveniles), which could indicate a relatively large number of individuals around the same age may mature in the near future (TEWG 2009). In-water studies throughout the eastern U.S.; however, indicate a substantial decrease in the abundance of the smallest oceanic/neritic juvenile loggerheads, a pattern corroborated by stranding data (TEWG 2009).

Population Estimate

The NMFS SEFSC developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS-SEFSC 2009). The model uses the range of published information for the various parameters including mortality by stage, stage duration (years in a stage), and fecundity parameters such as eggs per nest, nests per nesting female, hatchling emergence success, sex ratio, and remigration interval. Resulting trajectories of model runs for each individual recovery unit, and the western North Atlantic population as a whole, were found to be very similar. The model run estimates from the adult female population size for the western North Atlantic (from the 2004-2008 time frame), suggest the adult female population size is approximately 20,000-40,000 individuals, with a low likelihood of females' numbering up to 70,000 (NMFS-SEFSC 2009). A less robust estimate for total benthic females in the western North Atlantic was also obtained, yielding approximately 30,000-300,000 individuals, up to less than 1 million (NMFS-SEFSC 2009). A preliminary regional abundance survey of loggerheads within the northwestern Atlantic continental shelf for positively identified loggerhead in all strata estimated about 588,000 loggerheads (interquartile range of 382,000-817,000). When correcting for unidentified turtles in proportion to the ratio of identified turtles, the estimate increased to about 801,000 loggerheads (interquartile range of 521,000-1,111,000) (SEFSC 2011).

Threats (Specific to Loggerhead Sea Turtles)

The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 3.3.1. Yet the impact of fishery interactions is a point of further emphasis for this species. The joint NMFS and USFWS 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).

Regarding the impacts of pollution, loggerheads may be particularly affected by organochlorine contaminants; they have the highest organochlorine concentrations (Storelli et al. 2008a) and metal loads (D'Ilio et al. 2011) in sampled tissues among the sea turtle species. It is thought that dietary preferences were likely to be the main differentiating factor among sea turtle species. Storelli et al. (2008a) analyzed tissues from stranded loggerhead sea turtles and found that mercury accumulates in sea turtle livers while

cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991b).

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (DWH Trustees 2015). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

Unlike Kemp's ridleys, the majority of nesting for the Northwest Atlantic Ocean loggerhead DPS occurs on the Atlantic coast, and thus loggerheads were impacted to a relatively lesser degree. However, it is likely that impacts to the NGMRU of the NWA loggerhead DPS would be proportionally much greater than the impacts occurring to other recovery units. Impacts to nesting and oiling effects on a large proportion of the NGMRU recovery unit, especially mating and nesting adults likely had an impact on the NGMRU. Based on the response injury evaluations for Florida Panhandle and Alabama nesting beaches (which fall under the NFMRU), the Trustees estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH oil spill response activities on nesting beaches. Although the long-term effects remain unknown, the DWH oil spill event impacts to the Northern Gulf of Mexico Recovery Unit may result in some nesting declines in the future due to a large reduction of oceanic age classes during the DWH oil spill event. Although adverse impacts occurred to loggerheads, the proportion of the population that is expected to have been exposed to and directly impacted by the DWH oil spill event is relatively low. Thus we do not believe a population-level impact occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

Specific information regarding potential climate change impacts on loggerheads is also available. Modeling suggests an increase of 2°C in air temperature would result in a sex ratio of over 80% female offspring for loggerheads nesting near Southport, North Carolina. The same increase in air temperatures at nesting beaches in Cape Canaveral, Florida, would result in close to 100% female offspring. Such highly skewed sex ratios could undermine the reproductive capacity of the species. More ominously, an air temperature increase of 3°C is likely to exceed the thermal threshold of most nests, leading to egg mortality (Hawkes et al. 2007). Warmer sea surface temperatures have also been correlated with an earlier onset of loggerhead nesting in the spring (Hawkes et al. 2007;

Weishampel et al. 2004), short inter-nesting intervals (Hays et al. 2002), and shorter nesting seasons (Pike et al. 2006).

3.3.3 Leatherback Sea Turtles

The leatherback sea turtle was listed as endangered throughout its entire range on June 2, 1970, (35 FR 8491) under the Endangered Species Conservation Act of 1969.

Species Description and Distribution

The leatherback is the largest sea turtle in the world, with a curved carapace length (CCL) that often exceeds 5 ft (150 cm) and front flippers that can span almost 9 ft (270 cm) (NMFS and USFWS 1998b). Mature males and females can reach lengths of over 6 ft (2 m) and weigh close to 2,000 lb (900 kg). The leatherback does not have a bony shell. Instead, its shell is approximately 1.5 in (4 cm) thick and consists of a leathery, oil-saturated connective tissue overlaying loosely interlocking dermal bones. The ridged shell and large flippers help the leatherback during its long-distance trips in search of food.

Unlike other sea turtles, leatherbacks have several unique traits that enable them to live in cold water. For example, leatherbacks have a countercurrent circulatory system (Greer et al. 1973),¹¹ a thick layer of insulating fat (Davenport et al. 1990; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),¹² and they can increase their body temperature through increased metabolic activity (Bostrom and Jones 2007; Southwood et al. 2005). These adaptations allow leatherbacks to be comfortable in a wide range of temperatures, which helps them to travel further than any other sea turtle species (NMFS and USFWS 1995). For example, a leatherback may swim more than 6,000 miles (10,000 km) in a single year (Benson et al. 2007a; Benson et al. 2011; Eckert 2006; Eckert et al. 2006). They search for food between latitudes 71°N and 47°S in all oceans, and travel extensively to and from their tropical nesting beaches. In the Atlantic Ocean, leatherbacks have been recorded as far north as Newfoundland, Canada, and Norway, and as far south as Uruguay, Argentina, and South Africa (NMFS 2001b).

While leatherbacks will look for food in coastal waters, they appear to prefer the open ocean at all life stages (Heppell et al. 2003b). Leatherbacks have pointed tooth-like cusps and sharp-edged jaws that are adapted for a diet of soft-bodied prey such as jellyfish and salps. A leatherback's mouth and throat also have backward-pointing spines that help retain jelly-like prey. Leatherbacks' favorite prey (e.g., jellyfish) occurs commonly in temperate and northern or sub-arctic latitudes and likely has a strong influence on leatherback distribution in these areas (Plotkin 2003). Leatherbacks are known to be deep divers, with recorded depths in excess of a half-mile (Eckert et al. 1989), but they may also come into shallow waters to locate prey items.

¹¹ Countercurrent circulation is a highly efficient means of minimizing heat loss through the skin's surface because heat is recycled. For example, a countercurrent circulation system often has an artery containing warm blood from the heart surrounded by a bundle of veins containing cool blood from the body's surface.

¹² "Gigantothermy" refers to a condition when an animal has relatively high volume compared to its surface area, and as a result, it loses less heat.

Genetic analyses using microsatellite markers along with mitochondrial DNA and tagging data indicate there are 7 groups or breeding populations in the Atlantic Ocean: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (TEWG 2007). General differences in migration patterns and foraging grounds may occur between the 7 nesting assemblages, although data to support this is limited in most cases.

Life History Information

The leatherback life cycle is broken into several stages: (1) egg/hatchling, (2) post-hatchling, (3) juvenile, (4) subadult, and (5) adult. Leatherbacks are a long-lived species that delay age of maturity, have low and variable survival in the egg and juvenile stages, and have relatively high and constant annual survival in the subadult and adult life stages (Chaloupka 2002; Crouse 1999; Heppell et al. 1999; Heppell et al. 2003b; Spotila et al. 1996; Spotila et al. 2000). While a robust estimate of the leatherback sea turtle's life span does not exist, the current best estimate for the maximum age is 43 (Avens et al. 2009). It is still unclear when leatherbacks first become sexually mature. Using skeletochronological data, Avens et al. (2009) estimated that leatherbacks in the western North Atlantic may not reach maturity until 29 years of age, which is longer than earlier estimates of 2-3 years by Pritchard and Trebbau (1984), of 3-6 years by Rhodin (1985), of 13-14 years for females by Zug and Parham (1996), and 12-14 years for leatherbacks nesting in the U.S. Virgin Islands by Dutton et al. (2005). A more recent study that examined leatherback growth rates estimated an age at maturity of 16.1 years (Jones et al. 2011). The average size of reproductively active females in the Atlantic is generally 5-5.5 ft (150-162 cm) CCL (Benson et al. 2007a; Hirth et al. 1993; Starbird and Suarez 1994). Still, females as small as 3.5-4 ft (105-125 cm) CCL have been observed nesting at various sites (Stewart et al. 2007).

Female leatherbacks typically nest on sandy, tropical beaches at intervals of 2-4 years (Garcia M. and Sarti 2000; McDonald and Dutton 1996; Spotila et al. 2000). Unlike other sea turtle species, female leatherbacks do not always nest at the same beach year after year; some females may even nest at different beaches during the same year (Dutton et al. 2005; Eckert 1989; Keinath and Musick 1993; Steyermark et al. 1996). Individual female leatherbacks have been observed with fertility spans as long as 25 years (Hughes 1996). Females usually lay up to 10 nests during the 3-6 month nesting season (March through July in the U.S.), typically 8-12 days apart, with 100 eggs or more per nest (Eckert et al. 2012; Eckert 1989; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). Yet, up to approximately 30% of the eggs may be infertile (Eckert 1989; Eckert et al. 1984; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50% worldwide (Eckert et al. 2012), which is lower than the greater than 80% reported for other sea turtle species (Miller 1997). In the U.S., the emergent success is higher at 54-72% (Eckert and Eckert 1990; Stewart and Johnson 2006; Tucker 1988). Thus the number of hatchlings in a given year may be less than the total number of eggs produced in a season. Eggs hatch after 60-65 days, and the hatchlings have white striping along the ridges of their backs and on the edges of the flippers. Leatherback hatchlings weigh approximately 1.5-2 oz. (40-50 g), and have length of approximately 2-3 in (51-76

mm), with fore flippers as long as their bodies. Hatchlings grow rapidly with reported growth rates for leatherbacks from 2.5-27.6 in (6-70 cm) in length, estimated at 12.6 in (32 cm) per year (Jones et al. 2011).

In the Atlantic, the sex ratio appears to be skewed toward females. The Turtle Expert Working Group (TEWG) reports that nearshore and onshore strandings data from the U.S. Atlantic and Gulf of Mexico coasts indicate that 60% of strandings were females (TEWG 2007). Those data also show that the proportion of females among adults (57%) and juveniles (61%) was also skewed toward females in these areas (TEWG 2007). James et al. (2007) collected size and sex data from large subadult and adult leatherbacks off Nova Scotia and also concluded a bias toward females at a rate of 1.86:1.

The survival and mortality rates for leatherbacks are difficult to estimate and vary by location. For example, the annual mortality rate for leatherbacks that nested at Playa Grande, Costa Rica, was estimated to be 34.6% in 1993-1994, and 34.0% in 1994-1995 (Spotila et al. 2000). In contrast, leatherbacks nesting in French Guiana and St. Croix had estimated annual survival rates of 91% (Rivalan et al. 2005) and 89% (Dutton et al. 2005), respectively. For the St. Croix population, the average annual juvenile survival rate was estimated to be approximately 63% and the total survival rate from hatchling to first year of reproduction for a female was estimated to be between 0.4% and 2%, assuming age at first reproduction is between 9-13 years (Eguchi et al. 2006). Spotila et al. (1996) estimated first-year survival rates for leatherbacks at 6.25%.

Migratory routes of leatherbacks are not entirely known; however, recent information from satellite tags have documented long travels between nesting beaches and foraging areas in the Atlantic and Pacific Ocean basins (Benson et al. 2007a; Benson et al. 2011; Eckert 2006; Eckert et al. 2006; Ferraroli et al. 2004; Hays et al. 2004; James et al. 2005). Leatherbacks nesting in Central America and Mexico travel thousands of miles through tropical and temperate waters of the South Pacific (Eckert and Sarti 1997; Shillinger et al. 2008). Data from satellite tagged leatherbacks suggest that they may be traveling in search of seasonal aggregations of jellyfish (Benson et al. 2007b; Bowlby et al. 1994; Graham 2009; Shenker 1984; Starbird et al. 1993; Suchman and Brodeur 2005)

Status and Population Dynamics

The status of the Atlantic leatherback population has been less clear than the Pacific population, which has shown dramatic declines at many nesting sites (Santidrián Tomillo et al. 2007; Sarti Martínez et al. 2007; Spotila et al. 2000). This uncertainty resulted from inconsistent beach and aerial surveys, cycles of erosion, and reformation of nesting beaches in the Guianas (representing the largest nesting area). Leatherbacks also show a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species. Coordinated efforts of data collection and analyses by the leatherback TEWG have helped to clarify the understanding of the Atlantic population status up through the early 2000's (TEWG 2007). However, additional information for the Northwest Atlantic population has more recently shown declines in that population as well, contrary to what earlier information indicated (Northwest Atlantic Leatherback Working Group 2018). A full status review covering leatherback status and trends for all populations worldwide is being finalized (2019).

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (TEWG 2007). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with most of the nesting occurring in the Guianas and Trinidad. The Southern Caribbean/Guianas stock of leatherbacks was designated after genetics studies indicated that animals from the Guianas (and possibly Trinidad) should be viewed as a single population. Using nesting females as a proxy for population, the TEWG (2007) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate. TEWG observed positive growth within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007). More specifically, Tiwari et al. (2013) report an estimated three-generation abundance change of +3%, +20,800%, +1,778%, and +6% in Trinidad, Guyana, Suriname, and French Guiana, respectively. However, subsequent analysis using data up through 2017 has shown decreases in this stock, with an annual geometric mean decline of 10.43% over what they described as the short term (2008-2017) and a long-term (1990-2017) annual geometric mean decline of 5% (Northwest Atlantic Leatherback Working Group 2018).

Researchers believe the cyclical pattern of beach erosion and then reformation has affected leatherback nesting patterns in the Guianas. For example, between 1979 and 1986, the number of leatherback nests in French Guiana had increased by about 15% annually (NMFS 2001b). This increase was then followed by a nesting decline of about 15% annually. This decline corresponded with the erosion of beaches in French Guiana and increased nesting in Suriname. This pattern suggests that the declines observed since 1987 might actually be a part of a nesting cycle that coincides with cyclic beach erosion in Guiana (Schulz 1975). Researchers think that the cycle of erosion and reformation of beaches may have changed where leatherbacks nest throughout this region. The idea of shifting nesting beach locations was supported by increased nesting in Suriname,¹³ while the number of nests was declining at beaches in Guiana (Hilterman et al. 2003). Though this information suggested the long-term trend for the overall Suriname and French Guiana population was increasing. A more recent cycle of nesting declines from 2008-2017, as high as 31% annual decline in the Awala-Yalimapo area of French Guiana and almost 20% annual declines in Guyana, has changed the long-term nesting trends in the region negative as described above (Northwest Atlantic Leatherback Working Group 2018).

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. Across the Western Caribbean, nesting is most prevalent in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coastline of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from index nesting beaches in Tortuguero, Gandoca, and Pacuaré in Costa Rica indicate that the nesting population likely was not growing over the 1995-2005 time series (TEWG 2007). Other modeling of the nesting data for Tortuguero indicates a possible 67.8% decline between 1995 and 2006 (Troëng et al. 2007). Wallace et al. (2014) report an estimated three-generation abundance change of -72%, -24%, and +6% for Tortuguero, Gandoca, and Pacuare, respectively. Further decline of almost 6% annual geometric mean from 2008-

¹³ Leatherback nesting in Suriname increased by more than 10,000 nests per year since 1999 with a peak of 30,000 nests in 2001.

2017 reflects declines in nesting beaches throughout this stock (Northwest Atlantic Leatherback Working Group 2018).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, St. Croix (U.S. Virgin Islands), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (TEWG 2007). Wallace et al. (2014) report an estimated three-generation abundance change of -4% and +5,583% at Culebra and Fajardo, respectively. At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has varied from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (TEWG 2007). From 2006-2010, Wallace et al. (2014) report an annual growth rate of +7.5% in St. Croix and a three-generation abundance change of +1,058%. Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2% between 1994 and 2004 (TEWG 2007). The nesting trend reversed course later, with an annual geometric mean decline of 10% from 2008-2017 driving the long-term trend (1990-2017) down to a 2% annual decline (Northwest Atlantic Leatherback Working Group 2018).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. FWC Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 3.5). A similar pattern was also observed statewide (Table 3.6). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting. Overall, the trend shows growth on Florida's east coast beaches. Wallace et al. (2014) report an annual growth rate of 9.7% and a three-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, with a partial rebound in 2018. The annual geometric mean trend for Florida has been a decline of almost 7% from 2008-2017, but the long-term trend (1990-2017) remains positive with an annual geometric mean increase of over 9% (Northwest Atlantic Leatherback Working Group 2018).

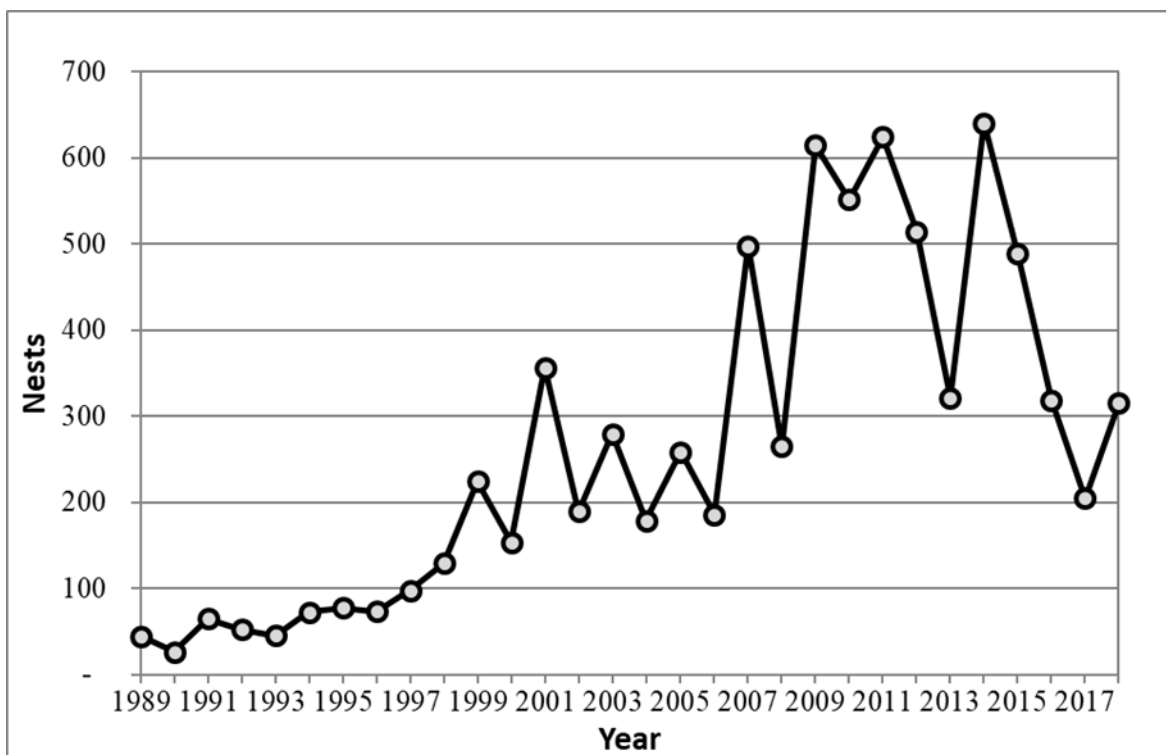


Figure 3.5. Leatherback sea turtle nesting at Florida index beaches since 1989

Table 3.6 Number of Leatherback Sea Turtle Nests in Florida

Nests Recorded	2011	2012	2013	2014	2015	2016	2017	2018
Index Nesting Beaches	625	515	322	641	489	319	205	316
Statewide	1,653	1,712	896	1,604	1,493	1,054	663	949

The West African nesting stock of leatherbacks is large and important, but it is a mostly unstudied aggregation. Nesting occurs in various countries along Africa's Atlantic coast, but much of the nesting is undocumented and the data are inconsistent. Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in a single season (Fretey et al. 2007). Fretey et al. (2007) provide detailed information about other known nesting beaches and survey efforts along the Atlantic African coast. Because of the lack of consistent effort and minimal available data, trend analyses were not possible for this stock (TEWG 2007).

Two other small but growing stocks nest on the beaches of Brazil and South Africa. Based on the data available, there was a positive annual average growth rate between 1.07% and 1.08% from 1988 and 2003 for the Brazilian stock and an estimated annual average growth rate between 1.04% and 1.06% for the South African stock (TEWG 2007).

Because the available nesting information is inconsistent, it is difficult to estimate the total population size for Atlantic leatherbacks. Spotila et al. (1996) characterized the entire Western Atlantic population as stable at best and estimated a population of 18,800 nesting females. Spotila et al. (1996) further estimated that the adult female leatherback

population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, was about 27,600 (considering both nesting and interesting females), with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the TEWG (2007). The TEWG (2007) also determined that at the time of their publication, leatherback sea turtle populations in the Atlantic were all stable or increasing with the exception of the Western Caribbean and West Africa populations. A later review by NMFS and USFWS (2013) suggested that the leatherback nesting population was stable in most nesting regions of the Atlantic Ocean. However, as described earlier, the NW Atlantic population has experienced declines over the near term (2008-2017), often severe enough to reverse the longer term trends to negative where increases had previously been seen (Northwest Atlantic Leatherback Working Group 2018). Given the relatively large size of the NW Atlantic population, it is likely that the overall Atlantic leatherback trend is no longer increasing.

Threats

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.2; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

Of all sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, especially gillnet and pot/trap lines. This vulnerability may be because of their body type (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, their method of locomotion, and/or their attraction to the lightsticks used to attract target species in longline fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine and many other stranded individuals exhibited evidence of prior entanglement (Dwyer et al. 2003). Zug and Parham (1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment from intense egg harvesting in some areas has caused a sharp decline in leatherback sea turtle populations and represents a significant threat to survival and recovery of the species worldwide.

Leatherback sea turtles may also be more susceptible to marine debris ingestion than other sea turtle species due to their predominantly pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding and migratory purposes (Lutcavage et al. 1997; Shoop and Kenney 1992). The stomach contents of leatherback sea turtles revealed that a substantial percentage (33.8% or 138 of 408 cases examined) contained some form of plastic debris (Mrosovsky et al. 2009). Blocking of the gut by plastic to an extent that could have caused death was evident in 8.7% of all leatherbacks that ingested plastic (Mrosovsky et al. 2009). Mrosovsky et al.

(2009) also note that in a number of cases, the ingestion of plastic may not cause death outright, but could cause the animal to absorb fewer nutrients from food, eat less in general, etc. - factors which could cause other adverse effects. The presence of plastic in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and forms of debris such as plastic bags (Mrosovsky et al. 2009). Balazs (1985a) speculated that the plastic object might resemble a food item by its shape, color, size, or even movement as it drifts about, and therefore induce a feeding response in leatherbacks.

As discussed in Section 3.2.2, global climate change can be expected to have various impacts on all sea turtles, including leatherbacks. Global climate change is likely to also influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007d). Several studies have shown leatherback distribution is influenced by jellyfish abundance (e.g., (Houghton et al. 2006; Witt et al. 2007; Witt et al. 2006); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH oil spill on leatherback sea turtles are considered here. Available information indicates leatherback sea turtles (along with hawksbill turtles) were likely least directly affected by the oil spill. Leatherbacks were documented in the spill area, but the number of affected leatherbacks was not estimated due to a lack of information compared to other species. But given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (TEWG 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, the Trustees conclude that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. (After the DWH oil spill, federal and state agencies came together to form the Deepwater Horizon Natural Resource Damage Assessment Trustee Council (“Trustees”). The Council studied the effects of the oil spill and continues to restore the Gulf of Mexico to the condition it would have been in if the spill had not happened.) Potential DWH-related impacts to leatherback sea turtles include direct oiling or contact with dispersants from surface and subsurface oil and dispersants, inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts likely occurred to leatherbacks, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event may be relatively low. Thus, a population-level impact may not have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

3.3.4 Kemp’s Ridley Sea Turtle

The Kemp’s ridley sea turtle was listed as endangered on December 2, 1970, under the Endangered Species Conservation Act of 1969, a precursor to the ESA. Internationally, the Kemp’s ridley is considered the most endangered sea turtle (Groombridge 1982; TEWG 2000a; Zwinenberg 1977).

Species Description and Distribution

The Kemp's ridley sea turtle is the smallest of all sea turtles. Adults generally weigh less than 100 lb (45 kg) and have a carapace length of around 2.1 ft (65 cm). Adult Kemp's ridley shells are almost as wide as they are long. Coloration changes significantly during development from the grey-black dorsum and plastron of hatchlings, a grey-black dorsum with a yellowish-white plastron as post-pelagic juveniles, and then to the lighter grey-olive carapace and cream-white or yellowish plastron of adults. There are 2 pairs of prefrontal scales on the head, 5 vertebral scutes, usually 5 pairs of costal scutes, and generally 12 pairs of marginal scutes on the carapace. In each bridge adjoining the plastron to the carapace, there are 4 scutes, each of which is perforated by a pore.

Kemp's ridley habitat largely consists of sandy and muddy areas in shallow, nearshore waters less than 120 ft (37 m) deep, although they can also be found in deeper offshore waters. These areas support the primary prey species of the Kemp's ridley sea turtle, which consist of swimming crabs, but may also include fish, jellyfish, and an array of mollusks.

The primary range of Kemp's ridley sea turtles is within the Gulf of Mexico basin, though they also occur in coastal and offshore waters of the U.S. Atlantic Ocean. Juvenile Kemp's ridley sea turtles, possibly carried by oceanic currents, have been recorded as far north as Nova Scotia. Historic records indicate a nesting range from Mustang Island, Texas, in the north to Veracruz, Mexico, in the south. Kemp's ridley sea turtles have recently been nesting along the Atlantic Coast of the U.S., with nests recorded from beaches in Florida, Georgia, and the Carolinas. In 2012, the first Kemp's ridley sea turtle nest was recorded in Virginia. The Kemp's ridley nesting population had been exponentially increasing prior to the recent low nesting years, which may indicate that the population had been experiencing a similar increase. Additional nesting data in the coming years will be required to determine what the recent nesting decline means for the population trajectory.

Life History Information

Kemp's ridley sea turtles share a general life history pattern similar to other sea turtles. Females lay their eggs on coastal beaches where the eggs incubate in sandy nests. After 45-58 days of embryonic development, the hatchlings emerge and swim offshore into deeper, ocean water where they feed and grow until returning at a larger size. Hatchlings generally range from 1.65-1.89 in (42-48 mm) straight carapace length (SCL), 1.26-1.73 in (32-44 mm) in width, and 0.3-0.4 lb (15-20 g) in weight. Their return to nearshore coastal habitats typically occurs around 2 years of age (Ogren 1989a), although the time spent in the oceanic zone may vary from 1-4 years or perhaps more (TEWG 2000). Juvenile Kemp's ridley sea turtles use these nearshore coastal habitats from April through November, but they move towards more suitable overwintering habitat in deeper offshore waters (or more southern waters along the Atlantic coast) as water temperature drops.

The average rates of growth may vary by location, but generally fall within $2.2\text{-}2.9 \pm 2.4$ in per year ($5.5\text{-}7.5 \pm 6.2$ cm/year) (Schmid and Barichivich 2006; Schmid and Woodhead 2000). Age to sexual maturity ranges greatly from 5-16 years, though NMFS et al. (2011a) determined the best estimate of age to maturity for Kemp's ridley sea turtles was 12 years. It is unlikely that most adults grow very much after maturity. While some sea turtles nest

annually, the weighted mean remigration rate for Kemp's ridley sea turtles is approximately 2 years. Nesting generally occurs from April to July. Females lay approximately 2.5 nests per season with each nest containing approximately 100 eggs (Márquez M. 1994).

Population Dynamics

Of the 7 species of sea turtles in the world, the Kemp's ridley has declined to the lowest population level. Most of the population of adult females nest on the beaches of Rancho Nuevo, Mexico (Pritchard 1969). When nesting aggregations at Rancho Nuevo were discovered in 1947, adult female populations were estimated to be in excess of 40,000 individuals (Hildebrand 1963). By the mid-1980s, however, nesting numbers from Rancho Nuevo and adjacent Mexican beaches were below 1,000, with a low of 702 nests in 1985. Yet, nesting steadily increased through the 1990s, and then accelerated during the first decade of the twenty-first century (Figure 3.6), which indicates the species is recovering.

It is worth noting that when the Bi-National Kemp's Ridley Sea Turtle Population Restoration Project was initiated in 1978, only Rancho Nuevo nests were recorded. In 1988, nesting data from southern beaches at Playa Dos and Barra del Tordo were added. In 1989, data from the northern beaches of Barra Ostionales and Tepehuajes were added, and most recently in 1996, data from La Pesca and Altamira beaches were recorded. Currently, nesting at Rancho Nuevo accounts for just over 81% of all recorded Kemp's ridley nests in Mexico. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley nests in Mexico reached a record high of 21,797 in 2012 (Gladys Porter Zoo 2013). From 2013 through 2014, there was a second significant decline, as only 16,385 and 11,279 nests were recorded, respectively. More recent data, however, indicates an increase in nesting. In 2015 there were 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Gladys Porter Zoo 2016). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm., August 31, 2017), but nesting for 2018 has declined to 17,945 (Gladys Porter Zoo data presentation by J. Pena, 2018). At this time, it is unclear whether the increases and declines in nesting seen over the past decade represents a population oscillating around an equilibrium point or if nesting will decline or increase in the future.

A small nesting population is also emerging in the U.S., primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (National Park Service data, <http://www.nps.gov/pais/naturescience/strp.htm>, <http://www.nps.gov/pais/naturescience/current-season.htm>). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010 followed by a second decline in 2013-2014, but with a rebound in 2015.

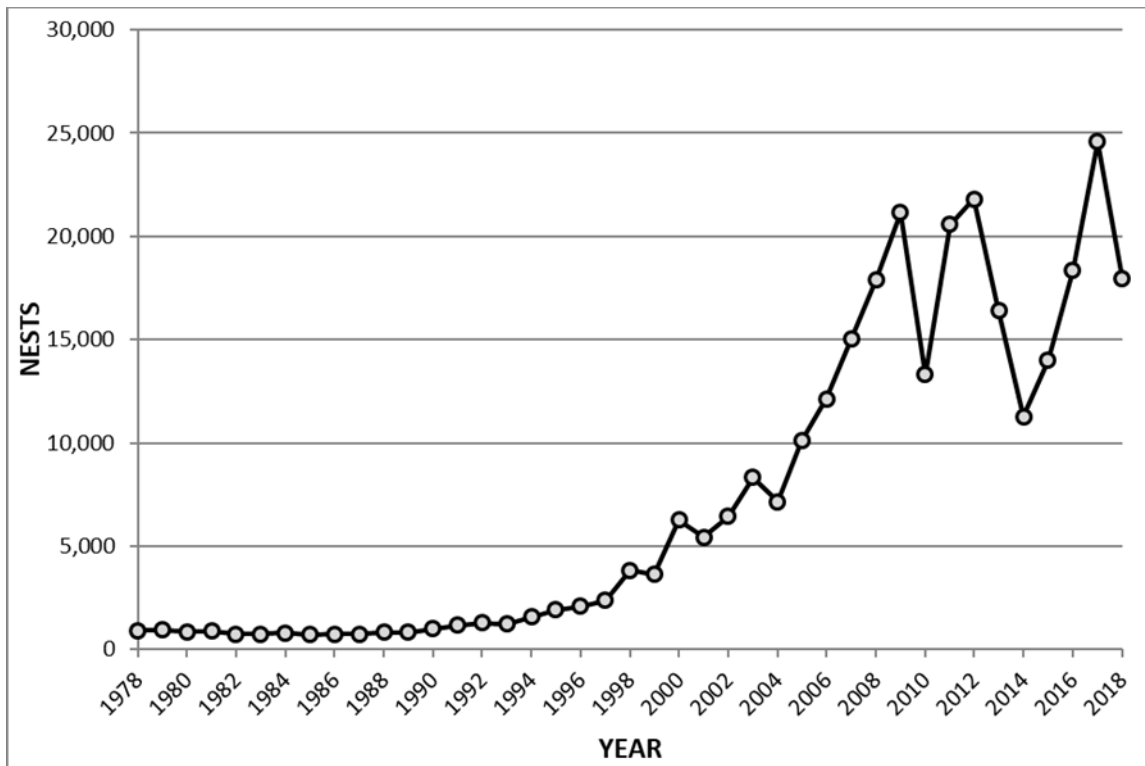


Figure 3.6 Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2016)

Through modelling, Heppell et al. (2005) predicted the population would increase at least 12-16% per year and could reach at least 10,000 females nesting on Mexico beaches by 2015. NMFS et al. (2011a) produced an updated model that predicted the population to increase 19% per year and to 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. While counts did not reach 25,000 nests by 2015, it is clear that the population has increased over the long term. The increases in Kemp's ridley sea turtle nesting over the last 2 decades is likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of TEDs, reduced trawling effort in Mexico and the U.S., and possibly other changes in vital rates (TEWG 1998a; TEWG 2000a). While these results are encouraging, the species' limited range as well as low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental randomness, all factors which are often difficult to predict with any certainty. Additionally, the significant nesting declines observed in 2010 and 2013-2014 potentially indicate a serious population-level impact, and there is cause for concern regarding the ongoing recovery trajectory.

Threats

Kemp's ridley sea turtles face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem

alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.3.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact Kemp's ridley sea turtles.

As Kemp's ridley sea turtles continue to recover and nesting arribadas¹⁴ are increasingly established, bacterial and fungal pathogens in nests are also likely to increase. Bacterial and fungal pathogen impacts have been well documented in the large arribadas of the olive ridley at Nancite in Costa Rica (Mo 1988). In some years, and on some sections of the beach, the hatching success can be as low as 5% (Mo 1988). As the Kemp's ridley nest density at Rancho Nuevo and adjacent beaches continues to increase, appropriate monitoring of emergence success will be necessary to determine if there are any density-dependent effects.

Since 2010, NMFS has documented (via the Sea Turtle Stranding and Salvage Network data, <http://www.sefsc.noaa.gov/species/turtles/strandings.htm>) elevated sea turtle strandings in the Northern Gulf of Mexico, particularly throughout the Mississippi Sound area. For example, in the first 3 weeks of June 2010, over 120 sea turtle strandings were reported from Mississippi and Alabama waters, none of which exhibited any signs of external oiling to indicate effects associated with the DWH oil spill event. A total of 644 sea turtle strandings were reported in 2010 from Louisiana, Mississippi, and Alabama waters, 561 (87%) of which were Kemp's ridley sea turtles. During March through May of 2011, 267 sea turtle strandings were reported from Mississippi and Alabama waters alone. A total of 525 sea turtle strandings were reported in 2011 from Louisiana, Mississippi, and Alabama waters, with the majority (455) having occurred from March through July, 390 (86%) of which were Kemp's ridley sea turtles. During 2012, a total of 384 sea turtles were reported from Louisiana, Mississippi, and Alabama waters. Of these reported strandings, 343 (89%) were Kemp's ridley sea turtles. During 2014, a total of 285 sea turtles were reported from Louisiana, Mississippi, and Alabama waters, though the data is incomplete. Of these reported strandings, 229 (80%) were Kemp's ridley sea turtles. These stranding numbers are significantly greater than reported in past years; Louisiana, Mississippi, and Alabama waters reported 42 and 73 sea turtle strandings for 2008 and 2009, respectively. It should be noted that stranding coverage has increased considerably due to the DWH oil spill event.

Nonetheless, considering that strandings typically represent only a small fraction of actual mortality, these stranding events potentially represent a serious impact to the recovery and survival of the local sea turtle populations. While a definitive cause for these strandings has not been identified, necropsy results indicate a significant number of stranded turtles from these events likely perished due to forced submergence, which is commonly associated with fishery interactions (B. Stacy, NMFS, pers. comm. to M. Barnette, NMFS SERO PRD, March 2012). Yet, available information indicates fishery effort was extremely limited during the stranding events. The fact that 80% or more of all Louisiana,

¹⁴ Arribada is the Spanish word for "arrival" and is the term used for massive synchronized nesting within the genus *Lepidochelys*.

Mississippi, and Alabama stranded sea turtles in the past 5 years were Kemp's ridleys is notable; however, this could simply be a function of the species' preference for shallow, inshore waters coupled with increased population abundance, as reflected in recent Kemp's ridley nesting increases.

In response to these strandings, and due to speculation that fishery interactions may be the cause, fishery observer effort was shifted to evaluate the inshore skimmer trawl fishery during the summer of 2012. During May-July of that year, observers reported 24 sea turtle interactions in the skimmer trawl fishery. All were identified as Kemp's ridleys but for a single sea turtle (1 sea turtle was an unidentified hardshell turtle). Encountered sea turtles were all very small juvenile specimens, ranging from 7.6-19.0 in (19.4-48.3 cm) curved carapace length (CCL). All sea turtles were released alive. The small average size of encountered Kemp's ridleys introduces a potential conservation issue, as over 50% of these reported sea turtles could potentially pass through the maximum 4-in bar spacing of TEDs currently required in the shrimp fishery. Due to this issue, a proposed 2012 rule to require TEDs in the skimmer trawl fishery (77 FR 27411) was not implemented. Following additional gear testing, NMFS proposed a new rule in 2016 to require TEDs with 3-in bar spacing for skimmer trawl vessels (81 FR 91097). Based on anecdotal information, these interactions were a relatively new issue for the inshore skimmer trawl fishery. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of Mexico may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles as analyzed in DWH Trustees (2016) are considered here. Kemp's ridleys experienced the greatest negative impact stemming from the DWH oil spill event of any sea turtle species. Impacts to Kemp's ridley sea turtles occurred to offshore small juveniles, as well as large juveniles and adults. Loss of hatchling production resulting from injury to adult turtles was also estimated for this species. Injuries to adult turtles of other species, such as loggerheads, certainly would have resulted in unrealized nests and hatchlings to those species as well. Yet, DWH Trustees (2016) only calculated unrealized nests and hatchlings of Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011a), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of Mexico throughout their lives (DWH Trustees 2016).

DWH Trustees (2016) estimated a total of 217,000 small juvenile Kemp's ridleys (51.5% of the total small juvenile sea turtle exposures to oil from the spill) were exposed to oil. That means approximately half of all small juvenile Kemp's ridleys from the total population estimate of 430,000 oceanic small juveniles were exposed to oil. Furthermore, a large number of small juveniles were removed from the population, DWH Trustee (2016) estimated up to 90,300 small juveniles Kemp's ridleys died as a direct result of the exposure. Therefore, as much as 20% of the small oceanic juveniles of this species were killed during that year. Impacts to large juveniles (>3 years old) and adults were also high.

An estimated 21,990 such individuals were exposed to oil (about 22% of the total estimated population for those age classes); of those, 3,110 mortalities were estimated (or 3% of the population for those age classes). The loss of near-reproductive and reproductive-stage females would have contributed to some extent to the decline in total nesting abundance observed between 2011 and 2014. The estimated number of unrealized Kemp's ridley nests is between 1,300 and 2,000, which translates to between approximately 65,000 and 95,000 unrealized hatchlings (DWH Trustees 2016). This is a minimum estimate, however, because the sublethal effects of the DWH oil spill event on turtles, their prey, and their habitats might have delayed or reduced reproduction in subsequent years, which may have contributed substantially to additional nesting deficits observed following the DWH oil spill event. These sublethal effects could have slowed growth and maturation rates, increased remigration intervals, and decreased clutch frequency (number of nests per female per nesting season). The nature of the DWH oil spill event effect on reduced Kemp's ridley nesting abundance and associated hatchling production after 2010 requires further evaluation. It is clear that the DWH oil spill event resulted in large losses to the Kemp's ridley population across various age classes, and likely had an important population-level effect on the species. Still, we do not have a clear understanding of those impacts on the population trajectory for the species into the future.

3.3.5 Green Sea Turtle (Information Relevant to All DPSs)

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978. The species was listed as threatened, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing determination was replaced with a listing determination applicable to 11 distinct population segments (DPSs) (81 FR 20057). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific were listed as threatened. For the purposes of this consultation, only the South Atlantic DPS (SA DPS) and North Atlantic DPS (NA DPS) will be considered, as they are the only two DPSs with individuals occurring in the Atlantic and Gulf of Mexico waters of the U.S.

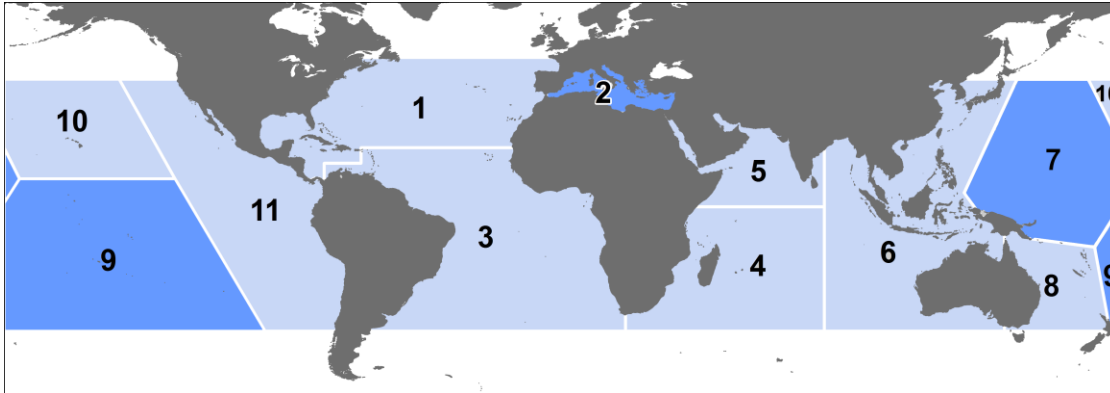


Figure 3.7 Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11. East Pacific.

Species Description and Distribution

The green sea turtle is the largest of the hardshell marine turtles, growing to a weight of 350 lb (159 kg) with a straight carapace length of greater than 3.3 ft (1 m). Green sea turtles have a smooth carapace with 4 pairs of lateral (or costal) scutes and a single pair of elongated prefrontal scales between the eyes. They typically have a black dorsal surface and a white ventral surface, although the carapace of green sea turtles in the Atlantic Ocean has been known to change in color from solid black to a variety of shades of grey, green, or brown and black in starburst or irregular patterns (Lagueux 2001).

With the exception of post-hatchlings, green sea turtles live in nearshore tropical and subtropical waters where they generally feed on marine algae and seagrasses. They have specific foraging grounds and may make large migrations between these forage sites and natal beaches for nesting (Hays et al. 2001). Green sea turtles nest on sandy beaches of mainland shores, barrier islands, coral islands, and volcanic islands in more than 80 countries worldwide (Hirth 1997). The 2 largest nesting populations are found at Tortuguero, on the Caribbean coast of Costa Rica (part of the NA DPS), and Raine Island, on the Pacific coast of Australia along the Great Barrier Reef.

Differences in mitochondrial DNA properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; FitzSimmons et al. 2006). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Within U.S. waters individuals from both the NA and SA DPSs can be found on foraging grounds. While there are currently no in-depth studies available to determine the percent of NA and SA DPS individuals in any given location, two small-scale studies provide an insight into the degree of mixing on the foraging grounds. An analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of Mexico), found approximately 4% of individuals came from nesting stocks in the SA DPS (specifically Suriname, Aves Island, Brazil, Ascension Island, and Guinea Bissau) (Foley et al. 2007). On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the Aves Island/Suriname nesting

assemblage, which is part of the SA DPS (Bass and Witzell 2000). All of the individuals in both studies were benthic juveniles. Available information on green turtle migratory behavior indicates that long distance dispersal is only seen for juvenile turtles. This suggests that larger adult-sized turtles return to forage within the region of their natal rookeries, thereby limiting the potential for gene flow across larger scales (Monzón-Argüello et al. 2010). While all of the mainland U.S. nesting individuals are part of the NA DPS, the U.S. Caribbean nesting assemblages are split between the NA and SA DPS. Nesters in Puerto Rico are part of the NA DPS, while those in the U.S. Virgin Islands are part of the SA DPS. We do not currently have information on what percent of individuals on the U.S. Caribbean foraging grounds come from which DPS.

North Atlantic DPS Distribution

The NA DPS boundary is illustrated in Figure 3.7. Four regions support nesting concentrations of particular interest in the NA DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

The complete nesting range of NA DPS green sea turtles within the southeastern U.S. includes sandy beaches between Texas and North Carolina, as well as Puerto Rico (Dow et al. 2007; NMFS and USFWS 1991). The vast majority of green sea turtle nesting within the southeastern U.S. occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Principal U.S. nesting areas for green sea turtles are in eastern Florida, predominantly Brevard south through Broward counties.

In U.S. Atlantic and Gulf of Mexico waters, green sea turtles are distributed throughout inshore and nearshore waters from Texas to Massachusetts. Principal benthic foraging areas in the southeastern U.S. include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984; Hildebrand 1982; Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1992; Wershoven and Wershoven 1992). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatán Peninsula.

South Atlantic DPS Distribution

The SA DPS boundary is shown in Figure 3.7, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

The in-water range of the SA DPS is widespread. In the eastern South Atlantic, significant sea turtle habitats have been identified, including green turtle feeding grounds in Corisco Bay, Equatorial Guinea/Gabon (Formia 1999); Congo; Mussulo Bay, Angola (Carr and Carr 1991); as well as Principe Island. Juvenile and adult green turtles utilize foraging areas throughout the Caribbean areas of the South Atlantic, often resulting in interactions with fisheries occurring in those same waters (Dow et al. 2007). Juvenile green turtles from multiple rookeries also frequently utilize the nearshore waters off Brazil as foraging grounds as evidenced from the frequent captures by fisheries (Lima et al. 2010; López-Barrera et al. 2012; Marcovaldi et al. 2009). Genetic analysis of green turtles on the foraging grounds off Ubatuba and Almofala, Brazil show mixed stocks coming primarily from Ascension, Suriname and Trindade as a secondary source, but also Aves, and even sometimes Costa Rica (North Atlantic DPS)(Naro-Maciel et al. 2007; Naro-Maciel et al. 2012). While no nesting occurs as far south as Uruguay and Argentina, both have important foraging grounds for South Atlantic green turtles (Gonzalez Carman et al. 2011; Lezama 2009; López-Mendilaharsu et al. 2006; Prosdocimi et al. 2012; Rivas-Zinno 2012).

Life History Information

Green sea turtles reproduce sexually, and mating occurs in the waters off nesting beaches and along migratory routes. Mature females return to their natal beaches (i.e., the same beaches where they were born) to lay eggs (Balazs 1982; Frazer and Ehrhart 1985) every 2-4 years while males are known to reproduce every year (Balazs 1983). In the southeastern U.S., females generally nest between June and September, and peak nesting occurs in June and July (Witherington and Ehrhart 1989b). During the nesting season, females nest at approximately 2-week intervals, laying an average of 3-4 clutches (Johnson and Ehrhart 1996). Clutch size often varies among subpopulations, but mean clutch size is approximately 110-115 eggs. In Florida, green sea turtle nests contain an average of 136 eggs (Witherington and Ehrhart 1989b). Eggs incubate for approximately 2 months before hatching. Hatchling green sea turtles are approximately 2 inches (5 cm) in length and weigh approximately 0.9 ounces (25 grams). Survivorship at any particular nesting site is greatly influenced by the level of man-made stressors, with the more pristine and less disturbed nesting sites (e.g., along the Great Barrier Reef in Australia) showing higher survivorship values than nesting sites known to be highly disturbed (e.g., Nicaragua) (Campell and Lagueux 2005; Chaloupka and Limpus 2005).

After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. This early oceanic phase remains one of the most poorly understood aspects of green sea turtle life history (NMFS and USFWS 2007a).

Green sea turtles exhibit particularly slow growth rates of about 0.4-2 inches (1-5 cm) per year (Green 1993), which may be attributed to their largely herbivorous, low-net energy diet (Bjorndal 1982). At approximately 8-10 inches (20-25 cm) carapace length, juveniles leave the pelagic environment and enter nearshore developmental habitats such as protected lagoons and open coastal areas rich in sea grass and marine algae. Growth studies using skeletochronology indicate that green sea turtles in the western Atlantic shift from the oceanic phase to nearshore developmental habitats after approximately 5-6 years (Bresette et al. 2006; Zug and Glor 1998). Within the developmental habitats, juveniles begin the switch to a more herbivorous diet, and by adulthood feed almost exclusively on seagrasses and algae (Rebel 1974), although some populations are known to also feed heavily on invertebrates (Carballo et al. 2002). Green sea turtles mature slowly, requiring 20-50 years to reach sexual maturity (Chaloupka and Musick 1997; Hirth 1997).

While in coastal habitats, green sea turtles exhibit site fidelity to specific foraging and nesting grounds, and it is clear they are capable of “homing in” on these sites if displaced (McMichael et al. 2003). Reproductive migrations of Florida green sea turtles have been identified through flipper tagging and/or satellite telemetry. Based on these studies, the majority of adult female Florida green sea turtles are believed to reside in nearshore foraging areas throughout the Florida Keys and in the waters southwest of Cape Sable, and some post-nesting turtles also reside in Bahamian waters as well (NMFS and USFWS 2007a).

Status and Population Dynamics

Accurate population estimates for marine turtles do not exist because of the difficulty in sampling turtles over their geographic ranges and within their marine environments. Nonetheless, researchers have used nesting data to study trends in reproducing sea turtles over time. A summary of nesting trends and nester abundance is provided in the most recent status review for the species (Seminoff et al. 2015), with information for each of the DPSs.

North Atlantic DPS

The NA DPS is the largest of the 11 green turtle DPSs, with an estimated nester abundance of over 167,000 adult females from 73 nesting sites. Overall this DPS is also the most data rich. Eight of the sites have high levels of abundance (i.e., <1000 nesters), located in Costa Rica, Cuba, Mexico, and Florida. All major nesting populations demonstrate long-term increases in abundance (Seminoff et al. 2015).

Tortuguero, Costa Rica, is by far the predominant nesting site, accounting for an estimated 79% of nesting for the DPS (Seminoff et al. 2015). Nesting at Tortuguero appears to have been increasing since the 1970's, when monitoring began. For instance, from 1971-1975 there were approximately 41,250 average annual emergences documented and this number increased to an average of 72,200 emergences from 1992-1996 (Bjorndal et al. 1999). Troëng and Rankin (2005) collected nest counts from 1999-2003 and also reported increasing trends in the population consistent with the earlier studies, with nest count data suggesting 17,402-37,290 nesting females per year (NMFS and USFWS 2007a). Modeling

by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually.

In the continental U.S., green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida where an estimated 200-1,100 females nest each year (Meylan et al. 1994; Weishampel et al. 2003). Occasional nesting has also been documented along the Gulf Coast of Florida (Meylan et al. 1995). Green sea turtle nesting is documented annually on beaches of North Carolina, South Carolina, and Georgia, though nesting is found in low quantities (nesting databases maintained on www.seaturtle.org).

In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 3.8). According to data collected from Florida's index nesting beach survey from 1989-2018, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 38,954 in 2017. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011, and a return to the trend of biennial peaks in abundance thereafter (Figure 3.8). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9%. Increases have been even more rapid in recent years.

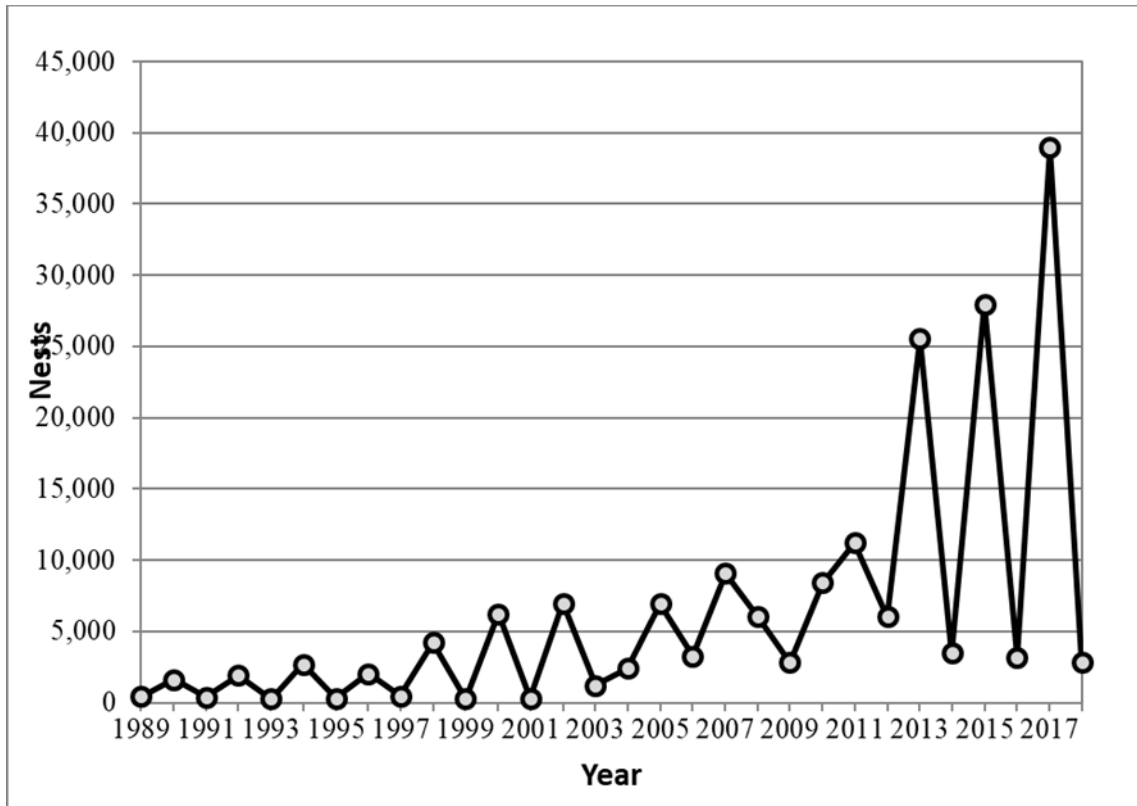


Figure 3.8 Green sea turtle nesting at Florida index beaches since 1989

Similar to the nesting trend found in Florida, in-water studies in Florida have also recorded increases in green turtle captures at the Indian River Lagoon site, with a 661% increase over 24 years (Ehrhart et al. 2007), and the St Lucie Power Plant site, with a significant increase in the annual rate of capture of immature green turtles (SCL<90 cm) from 1977 to 2002 or 26 years (3,557 green turtles total; M. Bressette, Inwater Research Group, unpubl. data; (Witherington et al. 2006).

South Atlantic DPS

The SA DPS is large, estimated at over 63,000 nesters, but data availability is poor. More than half of the 51 identified nesting sites (37) did not have sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). This includes some sites, such as beaches in French Guiana, which are suspected to have large numbers of nesters. Therefore, while the estimated number of nesters may be substantially underestimated, we also do not know the population trends at those data-poor beaches. However, while the lack of data was a concern due to increased uncertainty, the overall trend of the SA DPS was not considered to be a major concern as some of the largest nesting beaches such as Ascension Island, Aves Island (Venezuela), and Galibi (Suriname) appear to be increasing. Others such as Trindade (Brazil), Atol das Rocas (Brazil), and Poilão and the rest of Guinea-Bissau seem to be stable or do not have sufficient data to make a determination. Bioko (Equatorial Guinea) appears to be in decline but has less nesting than the other primary sites (Seminoff et al. 2015).

In the U.S., nesting of SA DPS green turtles occurs on the beaches of the U.S. Virgin Islands, primarily on Buck Island. There is insufficient data to determine a trend for Buck Island nesting, and it is a smaller rookery, with approximately 63 total nesters utilizing the beach (Seminoff et al. 2015).

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern U.S., green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.3.1.

In addition to general threats, green sea turtles are susceptible to natural mortality from Fibropapillomatosis (FP) disease. FP results in the growth of tumors on soft external tissues (flippers, neck, tail, etc.), the carapace, the eyes, the mouth, and internal organs (gastrointestinal tract, heart, lungs, etc.) of turtles (Aguirre et al. 2002; Herbst 1994; Jacobson et al. 1989). These tumors range in size from 0.04 inches (0.1 cm) to greater than 11.81 inches (30 cm) in diameter and may affect swimming, vision, feeding, and organ function (Aguirre et al. 2002; Herbst 1994; Jacobson et al. 1989). Presently, scientists are unsure of the exact mechanism causing this disease, though it is believed to be related to both an infectious agent, such as a virus (Herbst et al. 1995), and environmental conditions (e.g., habitat degradation, pollution, low wave energy, and shallow water (Foley et al. 2005)). FP is cosmopolitan, but it has been found to affect large numbers of animals in specific areas, including Hawaii and Florida (Herbst 1994; Jacobson 1990; Jacobson et al. 1991).

Cold-stunning is another natural threat to green sea turtles. Although it is not considered a major source of mortality in most cases, as temperatures fall below 46.4°-50°F (8°-10°C) turtles may lose their ability to swim and dive, often floating to the surface. The rate of cooling that precipitates cold-stunning appears to be the primary threat, rather than the water temperature itself (Milton and Lutz 2003). Sea turtles that overwinter in inshore waters are most susceptible to cold-stunning because temperature changes are most rapid in shallow water (Witherington and Ehrhart 1989a). During January 2010, an unusually large cold-stunning event in the southeastern U.S. resulted in around 4,600 sea turtles, mostly greens, found cold-stunned, and hundreds found dead or dying. A large cold-stunning event occurred in the western Gulf of Mexico in February 2011, resulting in approximately 1,650 green sea turtles found cold-stunned in Texas. Of these, approximately 620 were found dead or died after stranding, while approximately 1,030 turtles were rehabilitated and released. During this same time frame, approximately 340 green sea turtles were

found cold-stunned in Mexico, though approximately 300 of those were subsequently rehabilitated and released.

Whereas oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juvenile greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (DWH Trustees 2015). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

While green turtles regularly use the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic, and the proportion of the population using the northern Gulf of Mexico at any given time is relatively low. Although it is known that adverse impacts occurred and numbers of animals in the Gulf of Mexico were reduced as a result of the DWH oil spill, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event, as well as the impacts being primarily to smaller juveniles (lower reproductive value than adults and large juveniles), reduces the impact to the overall population. It is unclear what impact these losses may have caused on a population level, but it is not expected to have had a large impact on the population trajectory moving forward. However, recovery of green turtle numbers equivalent to what was lost in the northern Gulf of Mexico as a result of the spill will likely take decades of sustained efforts to reduce the existing threats and enhance survivorship of multiple life stages (DWH Trustees 2015).

3.3.6 Hawksbill Sea Turtle

The hawksbill sea turtle was listed as endangered throughout its entire range on June 2, 1970 (35 FR 8491), under the Endangered Species Conservation Act of 1969, a precursor to the ESA. Critical habitat was designated on June 2, 1998, in coastal waters surrounding Mona and Monito Islands in Puerto Rico (63 FR 46693).

Species Description and Distribution

Hawksbill sea turtles are small- to medium-sized (99-150 lb on average [45-68 kg]) although females nesting in the Caribbean are known to weigh up to 176 lb (80 kg) (Pritchard et al. 1983). The carapace is usually serrated and has a “tortoise-shell” coloring, ranging from dark to golden brown, with streaks of orange, red, and/or black. The plastron of a hawksbill turtle is typically yellow. The head is elongated and tapers to a point, with a beak-like mouth that gives the species its name. The shape of the mouth allows the hawksbill turtle to reach into holes and crevices of coral reefs to find sponges, their

primary adult food source, and other invertebrates. The shells of hatchlings are 1.7 in (42 mm) long, are mostly brown, and are somewhat heart-shaped (Eckert 1995; Hillis and Mackay 1989; van Dam and Sarti 1989).

Hawksbill sea turtles have a circumtropical distribution and usually occur between latitudes 30°N and 30°S in the Atlantic, Pacific, and Indian Oceans. In the western Atlantic, hawksbills 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 (Amos 1989; Groombridge and Luxmoore 1989; Lund 1985; Meylan and Donnelly 1999; NMFS and USFWS 1998a; Plotkin and Amos 1990; Plotkin and Amos 1988). They are highly migratory and use a wide range of habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Adult hawksbill sea turtles are capable of migrating long distances between nesting beaches and foraging areas. For instance, a female hawksbill sea turtle tagged at Buck Island Reef National Monument (BIRNM) in St. Croix was later identified 1,160 miles (1,866 km) away in the Miskito Cays in Nicaragua (Spotila 2004).

Hawksbill sea turtles nest on sandy beaches throughout the tropics and subtropics. Nesting occurs in at least 70 countries, although much of it now only occurs at low densities compared to that of other sea turtle species (NMFS and USFWS 2007b). Meylan and Donnelly (1999) believe that the widely dispersed nesting areas and low nest densities is likely a result of overexploitation of previously large colonies that have since been depleted over time. The most significant nesting within the U.S. occurs in Puerto Rico and the U.S. Virgin Islands, specifically on Mona Island and BIRNM, respectively. Although nesting within the continental U.S. is typically rare, it can occur along the southeast coast of Florida and the Florida Keys. The largest hawksbill nesting population in the western Atlantic occurs in the Yucatán Peninsula of Mexico, where several thousand nests are recorded annually in the states of Campeche, Yucatán, and Quintana Roo (Garduño-Andrade et al. 1999; Spotila 2004). In the U.S. Pacific, hawksbills nest on main island beaches in Hawaii, primarily along the east coast of the island. Hawksbill nesting has also been documented in American Samoa and Guam. More information on nesting in other ocean basins may be found in the 5-year status review for the species (NMFS and USFWS 2007b).

Mitochondrial DNA studies show that reproductive populations are effectively isolated over ecological time scales (Bass et al. 1996). Substantial efforts have been made to determine the nesting population origins of hawksbill sea turtles assembled in foraging grounds, and genetic research has shown that hawksbills of multiple nesting origins commonly mix in foraging areas (Bowen and Witzell 1996). Since hawksbill sea turtles nest primarily on the beaches where they were born, if a nesting population is decimated, it might not be replenished by sea turtles from other nesting rookeries (Bass et al. 1996).

Life History Information

Hawksbill sea turtles exhibit slow growth rates although they are known to vary within and among populations from a low of 0.4-1.2 in (1-3 cm) per year, measured in the Indo-Pacific (Chaloupka and Limpus 1997; Mortimer et al. 2003; Mortimer et al. 2002; Whiting

2000), to a high of 2 in (5 cm) or more per year, measured at some sites in the Caribbean (Diez and Van Dam 2002; León and Diez 1999). Differences in growth rates are likely due to differences in diet and/or density of sea turtles at foraging sites and overall time spent foraging (Bjorndal and Bolten 2002; Chaloupka et al. 2004). Consistent with slow growth, age to maturity for the species is also long, taking between 20 and 40 years, depending on the region (Chaloupka and Musick 1997; Limpus and Miller 2000). Hawksbills in the western Atlantic are known to mature faster (i.e., 20 or more years) than sea turtles found in the Indo-Pacific (i.e., 30-40 years) (Boulán 1983; Boulon Jr. 1994; Diez and Van Dam 2002; Limpus and Miller 2000). Males are typically mature when their length reaches 27 in (69 cm), while females are typically mature at 30 in (75 cm) (Eckert et al. 1992; Limpus 1992).

Female hawksbills return to the beaches where they were born (natal beaches) every 2-3 years to nest (Van Dam et al. 1991; Witzell 1983) and generally lay 3-5 nests per season (Richardson et al. 1999). Compared with other sea turtles, the number of eggs per nest (clutch) for hawksbills can be quite high. The largest clutches recorded for any sea turtle belong to hawksbills (approximately 250 eggs per nest) ((Hirth and Latif 1980), though nests in the U.S. Caribbean and Florida more typically contain approximately 140 eggs (USFWS hawksbill fact sheet, <http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm>). Eggs incubate for approximately 60 days before hatching (USFWS hawksbill fact sheet). Hatchling hawksbill sea turtles typically measure 1-2 in (2.5-5 cm) in length and weigh approximately 0.5 oz. (15 g).

Hawksbills may undertake developmental migrations (migrations as immatures) and reproductive migrations that involve travel over many tens to thousands of miles (Meylan 1999a). Post-hatchlings (oceanic stage juveniles) are believed to live in the open ocean, taking shelter in floating algal mats and drift lines of flotsam and jetsam in the Atlantic and Pacific oceans (Musick and Limpus 1997) before returning to more coastal foraging grounds. In the Caribbean, hawksbills are known to almost exclusively feed on sponges (Meylan 1988; Van Dam and Diez 1997), although at times they have been seen foraging on other food items, notably corallimorphs and zooanthids (León and Diez 2000; Mayor et al. 1998; Van Dam and Diez 1997).

Reproductive females undertake periodic (usually non-annual) migrations to their natal beaches to nest and exhibit a high degree of fidelity to their nest sites. Movements of reproductive males are less certain, but are presumed to involve migrations to nesting beaches or to courtship stations along the migratory corridor. Hawksbills show a high fidelity to their foraging areas as well (Van Dam and Diez 1998). Foraging sites are typically areas associated with coral reefs, although hawksbills are also found around rocky outcrops and high energy shoals which are optimum sites for sponge growth. They can also inhabit seagrass pastures in mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent (Bjorndal 1997; Van Dam and Diez 1998).

Status and Population Dynamics

There are currently no reliable estimates of population abundance and trends for non-nesting hawksbills at the time of this consultation; therefore, nesting beach data is currently the primary information source for evaluating trends in global abundance. Most hawksbill populations around the globe are either declining, depleted, and/or remnants of larger aggregations (NMFS and USFWS 2007b). The largest nesting population of hawksbills occurs in Australia where approximately 2,000 hawksbills nest off the northwest coast and about 6,000-8,000 nest off the Great Barrier Reef each year (Spotila 2004). Additionally, about 2,000 hawksbills nest each year in Indonesia and 1,000 nest in the Republic of Seychelles (Spotila 2004). In the U.S., hawksbills typically laid about 500-1,000 nests on Mona Island, Puerto Rico in the past (Diez and Van Dam 2007), but the numbers appear to be increasing, as the Puerto Rico Department of Natural and Environmental Resources counted nearly 1,600 nests in 2010 (PRDNER nesting data). Another 56-150 nests are typically laid on Buck Island off St. Croix (Meylan 1999b; Mortimer and Donnelly 2008). Nesting also occurs to a lesser extent on beaches on Culebra Island and Vieques Island in Puerto Rico, the mainland of Puerto Rico, and additional beaches on St. Croix, St. John, and St. Thomas, U.S. Virgin Islands.

Mortimer and Donnelly (2008) reviewed nesting data for 83 nesting concentrations organized among 10 different ocean regions (i.e., Insular Caribbean, Western Caribbean Mainland, Southwestern Atlantic Ocean, Eastern Atlantic Ocean, Southwestern Indian Ocean, Northwestern Indian Ocean, Central Indian Ocean, Eastern Indian Ocean, Western Pacific Ocean, Central Pacific Ocean, and Eastern Pacific Ocean). They determined historic trends (i.e., 20-100 years ago) for 58 of the 83 sites, and also determined recent abundance trends (i.e., within the past 20 years) for 42 of the 83 sites. Among the 58 sites where historic trends could be determined, all showed a declining trend during the long-term period. Among the 42 sites where recent (past 20 years) trend data were available, 10 appeared to be increasing, 3 appeared to be stable, and 29 appeared to be decreasing. With respect to regional trends, nesting populations in the Atlantic (especially in the Insular Caribbean and Western Caribbean Mainland) are generally doing better than those in the Indo-Pacific regions. For instance, 9 of the 10 sites that showed recent increases are located in the Caribbean. Buck Island and St. Croix's East End beaches support 2 remnant populations of between 17-30 nesting females per season (Hillis and Mackay 1989; Mackay 2006). While the proportion of hawksbills nesting on Buck Island represents a small proportion of the total hawksbill nesting occurring in the greater Caribbean region, Mortimer and Donnelly (2008) report an increasing trend in nesting at that site based on data collected from 2001-2006.

Nesting concentrations in the Pacific Ocean appear to be performing the worst of all regions despite the fact that the region currently supports more nesting hawksbills than either the Atlantic or Indian Oceans (Mortimer and Donnelly 2008). While still critically low in numbers, sightings of hawksbills in the eastern Pacific appear to have been increasing since 2007, though some of that increase may be attributable to better observations (Gaos et al. 2010). More information about site-specific trends can be found in the most recent 5-year status review for the species (NMFS and USFWS 2007b).

Threats

Hawksbills are currently subjected to the same suite of threats on both nesting beaches and in the marine environment that affect other sea turtles (e.g., interaction with federal and state fisheries, coastal construction, oil spills, climate change affecting sex ratios) as discussed in Section 3.2.2. There are also specific threats that are of special emphasis, or are unique, for hawksbill sea turtles discussed in further detail below.

While oil spill impacts are discussed generally for all species in Section 3.2.2, specific impacts of the DWH spill on hawksbill turtles have been estimated. Hawksbills made up 2.2% (8,850) of small juvenile sea turtle (of those that could be identified to species) exposures to oil in offshore areas, with an estimate of 615 to 3,090 individuals dying as a result of the direct exposure (DWH Trustees 2015). No quantification of large benthic juveniles or adults was made. Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts occurred to hawksbills, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event is relatively low, and thus a population-level impact is not believed to have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

The historical decline of the species is primarily attributed to centuries of exploitation for the beautifully patterned shell, which made it a highly attractive species to target (Parsons 1972). The fact that reproductive females exhibit a high fidelity for nest sites and the tendency of hawksbills to nest at regular intervals within a season made them an easy target for capture on nesting beaches. The shells from hundreds of thousands of sea turtles in the western Caribbean region were imported into the United Kingdom and France during the nineteenth and early twentieth centuries (Parsons 1972). Additionally, hundreds of thousands of sea turtles contributed to the region's trade with Japan prior to 1993 when a zero quota was imposed (Milliken and Tokunaga 1987), as cited in Brautigam and Eckert (2006).

The continuing demand for the hawksbills' shells as well as other products derived from the species (e.g., leather, oil, perfume, and cosmetics) represents an ongoing threat to its recovery. The British Virgin Islands, Cayman Islands, Cuba, Haiti, and the Turks and Caicos Islands (United Kingdom) all permit some form of legal take of hawksbill sea turtles. In the northern Caribbean, hawksbills continue to be harvested for their shells, which are often carved into hair clips, combs, jewelry, and other trinkets (Márquez M. 1990; Stapleton and Stapleton 2006). Additionally, hawksbills are harvested for their eggs and meat, while whole, stuffed sea turtles are sold as curios in the tourist trade. Hawksbill sea turtle products are openly available in the Dominican Republic and Jamaica, despite a prohibition on harvesting hawksbills and their eggs (Fleming 2001). Up to 500 hawksbills per year from 2 harvest sites within Cuba were legally captured each year until 2008 when the Cuban government placed a voluntary moratorium on the sea-turtle fishery (Carillo et

al. 1999; Mortimer and Donnelly 2008). While current nesting trends are unknown, the number of nesting females is suspected to be declining in some areas (Carillo et al. 1999; Moncada et al. 1999). International trade in the shell of this species is prohibited between countries that have signed the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), but illegal trade still occurs and remains an ongoing threat to hawksbill survival and recovery throughout its range.

Due to their preference to feed on sponges associated with coral reefs, hawksbill sea turtles are particularly sensitive to losses of coral reef communities. Coral reefs are vulnerable to destruction and degradation caused by human activities (e.g., nutrient pollution, sedimentation, contaminant spills, vessel groundings and anchoring, recreational uses) and are also highly sensitive to the effects of climate change (e.g., higher incidences of disease and coral bleaching) (Crabbe 2008; Wilkinson 2004). Because continued loss of coral reef communities (especially in the greater Caribbean region) is expected to impact hawksbill foraging, it represents a major threat to the recovery of the species.

3.3.7 Smalltooth Sawfish

The U.S. DPS of smalltooth sawfish was listed as endangered under the ESA effective May 1, 2003 (68 FR 15674, April 1, 2003).

Species Description and Distribution

The smalltooth sawfish is a tropical marine and estuarine elasmobranch. It has an extended snout with a long, narrow, flattened, rostral blade (rostrum) with a series of transverse teeth along either edge. In general, smalltooth sawfish inhabit shallow coastal waters of warm seas throughout the world and feed on a variety of small fish (e.g., mullet, jacks, and ladyfish) (Simpfendorfer 2001), and crustaceans (e.g., shrimp and crabs) (Bigelow and Schroeder 1953b; Norman and Fraser 1937).

Although this species is reported to have a circumtropical distribution, NMFS identified smalltooth sawfish from the Southeast U.S. as a distinct population segment (DPS), due to the physical isolation of this population from others, the differences in international management of the species, and the significance of the U.S. population in relation to the global range of the species (see 68 FR15674). Within the U.S., smalltooth sawfish have been captured in estuarine and coastal waters from New York southward through Texas, although peninsular Florida has historically been the region of the U.S. with the largest number of recorded captures (NMFS 2000). Recent records indicate there is a resident reproducing population of smalltooth sawfish in south and southwest Florida from Charlotte Harbor through the Dry Tortugas, which is also the last U.S. stronghold for the species (Poulakis and Seitz 2004a; Seitz and Poulakis 2002; Simpfendorfer and Wiley 2005a). Water temperatures (no lower than 16-18°C) and the availability of appropriate coastal habitat (shallow, euryhaline waters and red mangroves) are the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic. Most specimens captured along the Atlantic coast north of Florida are large adults (over 10 ft) that likely represent seasonal migrants, wanderers, or colonizers from a historic Florida core population(s) to the south, rather than being members of a continuous, even-density population (Bigelow and Schroeder 1953b).

Life History Information

Smalltooth sawfish fertilization is internal and females give birth to live young. The brood size, gestation period, and frequency of reproduction are unknown for smalltooth sawfish. Therefore, data from the closely related (in terms of size and body morphology) largetooth sawfish represent our best estimates of these parameters. The largetooth sawfish likely reproduces every other year, has a gestation period of approximately 5 months, and produces a mean of 7.3 offspring per brood, with a range of 1-13 offspring (Thorson 1976). Smalltooth sawfish are approximately 31 in (80 cm) at birth and may grow to a length of 18 ft (548 cm) or greater during their lifetime (Bigelow and Schroeder 1953b; Simpfendorfer 2002). Simpfendorfer et al. (2008) report rapid juvenile growth for smalltooth sawfish for the first 2 years after birth, with stretched total length increasing by an average of 25-33 in (65-85 cm) in the first year and an average of 19-27 in (48-68 cm) in the second year. By contrast, very little information exists on size classes other than juveniles, which make up the majority of sawfish encounters; therefore, much uncertainty remains in estimating life history parameters for smalltooth sawfish, especially as it relates to age at maturity and post-juvenile growth rates. Based on age and growth studies of the largetooth sawfish (Thorson 1982) and research by Simpfendorfer (2000), the smalltooth sawfish is likely a slow-growing (with the exception of early juveniles), late-maturing (10-20 years) species with a long lifespan (30-60 years). Juvenile growth rates presented by Simpfendorfer et al. (2008) suggest smalltooth sawfish are growing faster than previously thought and therefore may reach sexual maturity at an earlier age.

There are distinct differences in habitat use based on life history stage. Juvenile smalltooth sawfish, those up to 3 years of age or approximately 8 ft in length (Simpfendorfer et al. 2008), inhabit the shallow waters of estuaries and can be found in sheltered bays, dredged canals, along banks and sandbars, and in rivers (NMFS 2000). Juvenile smalltooth sawfish occur in euryhaline waters (i.e., waters with a wide range of salinities) and are often closely associated with muddy or sandy substrates, and shorelines containing red mangroves, *Rhizophora mangle* (Simpfendorfer 2001; Simpfendorfer 2003). Tracking data from the Caloosahatchee River in Florida indicate very shallow depths and salinity are important abiotic factors influencing juvenile smalltooth sawfish movement patterns, habitat use, and distribution (Simpfendorfer et al. 2011). Another recent acoustic tagging study in a developed region of Charlotte Harbor, Florida, identified the importance of mangroves in close proximity to shallow water habitat for juvenile smalltooth sawfish, stating that juveniles generally occur in shallow water within 328 ft (100 m) of mangrove shorelines, generally red mangroves (Simpfendorfer et al. 2010). Juvenile smalltooth sawfish spend the majority of their time in waters less than 13 ft (4 m) in depth (Simpfendorfer et al. 2010) and are seldom found in depths greater than 32 ft (10 m) (Poulakis and Seitz 2004a). Simpfendorfer et al. (2010) also indicated developmental differences in habitat use: the smallest juveniles (young-of-the-year juveniles measuring < 100 cm in length) generally used water depths less than 0.5 m (1.64 ft), had small home ranges (4,264-4,557 m²), and exhibited high levels of site fidelity. Although small juveniles exhibit high levels of site fidelity for specific nursery habitats for periods of time lasting up to 3 months (Wiley and Simpfendorfer 2007), they do undergo small movements coinciding with changing tidal stages. These movements often involve moving from shallow sandbars at low tide to within red mangrove prop roots at higher tides (Simpfendorfer et al. 2010), behavior likely

to reduce the risk of predation (Simpfendorfer 2006). As juveniles increase in size, they begin to expand their home ranges (Simpfendorfer et al. 2010; Simpfendorfer et al. 2011), eventually moving to more offshore habitats where they likely feed on larger prey and eventually reach sexual maturity.

Researchers have identified several areas within the Charlotte Harbor Estuary that are disproportionately more important to juvenile smalltooth sawfish, based on intra- or inter-annual (within or between year) capture rates during random sampling events within the estuary (Poulakis 2012; Poulakis et al. 2011). These areas were termed “hotspots” and also correspond with areas where public encounters are most frequently reported. Use of these “hotspots” can vary within and among years based on the amount and timing of freshwater inflow. Smalltooth sawfish use hotspots further upriver during high salinity conditions (drought) and areas closer to the mouth of the Caloosahatchee River during times of high freshwater inflow (Poulakis et al. 2011). At this time, researchers are unsure what specific biotic or abiotic factors influence this habitat use, but they believe a variety of conditions in addition to salinity, such as temperature, dissolved oxygen, water depth, shoreline vegetation, and food availability, may influence habitat selection (Poulakis et al. 2011).

While adult smalltooth sawfish may also use the estuarine habitats used by juveniles, they are commonly observed in deeper waters along the coasts. Poulakis and Seitz (2004a) noted that nearly half of the encounters with adult-sized smalltooth sawfish in Florida Bay and the Florida Keys occurred in depths from 200-400 ft (70-122 m) of water. Similarly, Simpfendorfer and Wiley (2005a) reported encounters in deeper waters off the Florida Keys, and observations from both commercial longline fishing vessels and fishery-independent sampling in the Florida Straits report large smalltooth sawfish in depths up to 130 ft (~40 m) (ISED 2014). Even so, NMFS believes adult smalltooth sawfish use shallow estuarine habitats during parturition (when adult females return to shallow estuaries to pup) because very young juveniles still containing rostral sheaths are captured in these areas. Since very young juveniles have high site fidelities, we hypothesize that they are birthed nearby or in their nursery habitats.

Status and Population Dynamics

Few long-term abundance data exist for the smalltooth sawfish, making it very difficult to estimate the current population size. Simpfendorfer (2001) estimated that the U.S. population may number less than 5% of historic levels, based on anecdotal data and the fact that the species’ range has contracted by nearly 90%, with south and southwest Florida the only areas known to support a reproducing population. Since actual abundance data are limited, researchers have begun to compile capture and sightings data (collectively referred to as encounter data) in the International Sawfish Encounter Database (ISED) that was developed in 2000. Although this data cannot be used to assess the population because of the opportunistic nature in which they are collected (i.e., encounter data are a series of random occurrences rather than an evenly distributed search over a defined period of time), researchers can use this database to assess the spatial and temporal distribution of smalltooth sawfish. We expect that as the population grows, the geographic range of encounters will also increase. Since the conception of the ISED, over 3,000 smalltooth

sawfish encounters have been reported and compiled in the encounter database (ISED 2014).

Despite the lack of scientific data on abundance, recent encounters with young-of-the-year, older juveniles, and sexually mature smalltooth sawfish indicate that the U.S. population is currently reproducing (Seitz and Poulakis 2002; Simpfendorfer 2003). The abundance of juveniles encountered, including very small individuals, suggests that the population remains viable (Simpfendorfer and Wiley 2004b), and data analyzed from Everglades National Park as part of an established fisheries-dependent monitoring program (angler interviews) indicate a slightly increasing trend in juvenile abundance within the park over the past decade (Carlson and Osborne 2012; Carlson et al. 2007). Using a demographic approach and life history data for smalltooth sawfish and similar species from the literature, Simpfendorfer (2000) estimated intrinsic rates of natural population increase for the species at 0.08-0.13 per year and population doubling times from 5.4-8.5 years. These low intrinsic rates¹⁵ of population increase, suggest that the species is particularly vulnerable to excessive mortality and rapid population declines, after which recovery may take decades.

Threats

Past literature indicates smalltooth sawfish were once abundant along both coasts of Florida and quite common along the shores of Texas and the northern Gulf coast (NMFS 2010) and citations therein). Based on recent comparisons with these historical reports, the U.S. DPS of smalltooth sawfish has declined over the past century (Simpfendorfer 2001; Simpfendorfer 2002). The decline in smalltooth sawfish abundance has been attributed to several factors including bycatch mortality in fisheries, habitat loss, and life history limitations of the species (NMFS 2010).

Bycatch Mortality

Bycatch mortality is cited as the primary cause for the decline in smalltooth sawfish in the U.S. (NMFS 2010). While there has never been a large-scale directed fishery, smalltooth sawfish easily become entangled in fishing gears (gill nets, otter trawls, trammel nets, and seines) directed at other commercial species, often resulting in serious injury or death (NMFS 2009). This has historically been reported in Florida (Snelson and Williams 1981), Louisiana (Simpfendorfer 2002), and Texas (Baughman 1943). For instance, one fisherman interviewed by Evermann and Bean (1897) reported taking an estimated 300 smalltooth sawfish in just one netting season in the Indian River Lagoon, Florida. In another example, smalltooth sawfish landings data gathered by Louisiana shrimp trawlers from 1945-1978, which contained both landings data and crude information on effort (number of vessels, vessel tonnage, number of gear units), indicated declines in smalltooth sawfish landings from a high of 34,900 lbs in 1949 to less than 1,500 lbs in most years after 1967. The Florida net ban passed in 1995 has led to a reduction in the number of smalltooth sawfish incidentally captured, "...by prohibiting the use of gill and other entangling nets in all Florida waters, and prohibiting the use of other nets larger than 500

¹⁵ The rate at which a population increases in size if there are no density-dependent forces regulating the population

square ft in mesh area in nearshore and inshore Florida waters”¹⁶ (FLA. CONST. art. X, § 16). However, the threat of bycatch currently remains in commercial fisheries (e.g., South Atlantic shrimp fishery, Gulf of Mexico shrimp fishery, federal shark fisheries of the South Atlantic, and the Gulf of Mexico reef fish fishery), though anecdotal information collected by NMFS port agents suggest smalltooth sawfish captures are now rare.

In addition to incidental bycatch in commercial fisheries, smalltooth sawfish have historically been and continue to be captured by recreational fishers. Encounter data (ISED 2014) and past research (Caldwell 1990) document that rostrums are sometimes removed from smalltooth sawfish caught by recreational fishers, thereby reducing their chances of survival. While the current threat of mortality associated with recreational fisheries is expected to be low given that possession of the species in Florida has been prohibited since 1992, bycatch in recreational fisheries remains a potential threat to the species.

Habitat Loss

Modification and loss of smalltooth sawfish habitat, especially nursery habitat, is another contributing factor in the decline of the species. Activities such as agricultural and urban development, commercial activities, dredge-and-fill operations, boating, erosion, and diversions of freshwater runoff contribute to these losses (SAFMC 1998). Large areas of coastal habitat were modified or lost between the mid-1970s and mid-1980s within the U.S. (Dahl and Johnson 1991). Since then, rates of loss have decreased, but habitat loss continues. From 1998-2004, approximately 64,560 acres of coastal wetlands were lost along the Atlantic and Gulf coasts of the U.S., of which approximately 2,450 acres were intertidal wetlands consisting of mangroves or other estuarine shrubs (Stedman and Dahl 2008). Further, Orlando et al. (1994) analyzed 18 major southeastern estuaries and recorded over 703 mi of navigation channels and 9,844 mi of shoreline with modifications. In Florida, coastal development often involves the removal of mangroves and the armoring of shorelines through seawall construction. Changes to the natural freshwater flows into estuarine and marine waters through construction of canals and other water control devices have had other impacts: altered the temperature, salinity, and nutrient regimes; reduced both wetlands and submerged aquatic vegetation; and degraded vast areas of coastal habitat utilized by smalltooth sawfish (Gilmore 1995; Reddering 1988; Whitfield and Bruton 1989). While these modifications of habitat are not the primary reason for the decline of smalltooth sawfish abundance, it is likely a contributing factor and almost certainly hampers the recovery of the species. Juvenile sawfish and their nursery habitats are particularly likely to be affected by these kinds of habitat losses or alternations, due to their affinity for shallow, estuarine systems. Although many forms of habitat modification are currently regulated, some permitted direct and/or indirect damage to habitat from increased urbanization still occurs and is expected to continue to threaten survival and recovery of the species in the future.

¹⁶ “nearshore and inshore Florida waters” means all Florida waters inside a line 3 mi seaward of the coastline along the Gulf of Mexico and inside a line 1 mi seaward of the coastline along the Atlantic Ocean.

Life History Limitations

The smalltooth sawfish is also limited by its life history characteristics as a slow-growing, relatively late-maturing, and long-lived species. Animals using this life history strategy are usually successful in maintaining small, persistent population sizes in constant environments, but are particularly vulnerable to increases in mortality or rapid environmental change (NMFS 2000). The combined characteristics of this life history strategy result in a very low intrinsic rate of population increase (Musick 1999) that make it slow to recover from any significant population decline (Simpfendorfer 2000). More recent data suggest smalltooth sawfish may mature earlier than previously thought, meaning rates of population increase could be higher and recovery times shorter than those currently reported (Simpfendorfer et al. 2008).

Current Threats

The 3 major factors that led to the current status of the U.S. DPS of smalltooth sawfish – bycatch mortality, habitat loss, and life history limitations – continue to be the greatest threats today. All the same, other threats such as the illegal commercial trade of smalltooth sawfish or their body parts, predation, and marine pollution and debris may also affect the population and recovery of smalltooth sawfish on smaller scales (NMFS 2010). We anticipate that all of these threats will continue to affect the rate of recovery for the U.S. DPS of smalltooth sawfish.

In addition to the anthropogenic effects mentioned previously, changes to the global climate are likely to be a threat to smalltooth sawfish and the habitats they use. The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal (IPCC 2007) and its impacts to coastal resources may be significant. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, changes in the amount and timing of precipitation, and changes in air and water temperatures (EPA 2012; NOAA 2012). The impacts to smalltooth sawfish cannot, for the most part, currently be predicted with any degree of certainty, but we can project some effects to the coastal habitats where they reside. We know that the coastal habitats that contain red mangroves and shallow, euryhaline waters will be directly impacted by climate change through sea level rise, which is expected to exceed 1 meter globally by 2100 according to Meehl et al. (2007), Pfeffer et al. (2008), and Vermeer and Rahmstorf (2009). Sea level rise will impact mangrove resources, as sediment surface elevations for mangroves will not keep pace with conservative projected rates of elevation in sea level (Gilman et al. 2008). Sea level increases will also affect the amount of shallow water available for juvenile smalltooth sawfish nursery habitat, especially in areas where there is shoreline armoring (e.g., seawalls). Further, the changes in precipitation coupled with sea level rise may also alter salinities of coastal habitats, reducing the amount of available smalltooth sawfish nursery habitat.

3.3.8 Atlantic Sturgeon

Species Descriptions and Distributions

Atlantic sturgeon are long-lived, late-maturing, estuarine-dependent, anadromous fish distributed along the eastern coast of North America (Waldman and Wirgin 1998).

Historically, sightings have been reported from Hamilton Inlet, Labrador, Canada, south to the St. Johns River, Florida (Murawski et al. 1977; Smith and Clugston 1997). Atlantic sturgeon may live up to 60 years, reach lengths up to 14 ft, and weigh over 800 lbs (ASSRT 2007; Collette and Klein-MacPhee 2002). They are distinguished by armor-like plates (called scutes) and a long protruding snout that has four barbels (slender, whisker-like feelers extending from the lower jaw used for touch and taste). Adult Atlantic sturgeon spend the majority of their lives in nearshore marine waters, returning to the rivers where they were born (natal rivers) to spawn (Wirgin et al. 2002). Young sturgeon may spend the first few years of life in their natal river estuary before moving out to sea (Wirgin et al. 2002). Atlantic sturgeon are omnivorous benthic (bottom) feeders and incidentally ingest mud along with their prey. Diets of adult and subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (ASSRT 2007; Bigelow and Schroeder 1953a; Guilbard et al. 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (ASSRT 2007; Bigelow and Schroeder 1953a; Guilbard et al. 2007).

Historically, Atlantic sturgeon were present in approximately 38 rivers in the United States from the St. Croix River, Maine to the St. Johns River, Florida, of which 35 rivers have been confirmed to have had a historical spawning subpopulation. Atlantic sturgeon are currently present in approximately 32 of these rivers, and spawning occurs in at least 20 of them. The marine range of Atlantic sturgeon extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida. The action area includes this range. The location of the action means subadult and adults could be effected by the action. Because adult and subadult Atlantic sturgeon from all DPSs mix extensively in marine waters, we expect fish from all DPSs to be found in the action area.

Life History Information

Atlantic sturgeon are generally referred to as having four size/developmental categories: larvae; young-of-year (YOY); juveniles and subadults; and adults. Because the action area occurs in marine waters where only subadult and adults are likely to occur, we will focus on those life stages here.

There is little morphometric difference between juveniles and subadults; they are distinguished by their occurrence within estuarine and marine waters. The term “juveniles” refers to animals 1 year of age or older that reside in the natal estuary. Juveniles are generally found in the lower estuaries near the freshwater/saltwater mixing zones but will move upriver and downriver within the natal estuary to remain in waters most suitable for their growth and development. As juveniles age and become larger, the range of habitat they can use expands to include a broader salinity range. Once suitably developed, juveniles make their first emigration from the natal river into the marine environment. There is some evidence to suggest this out migration of larger juveniles is influenced by the density of younger, less-developed juveniles. Because early juveniles are intolerant of salinity, they are likely unable to use foraging habitats in coastal waters if riverine food resources become limited. However, older, more-developed juveniles are able to use these coastal habitat, though they may prefer the relatively predator-free environments of brackish water estuaries as long as food resources are not limited (Schueller and Peterson 2010).

These movements into marine waters mark the beginning of the subadult stage. Thus, subadults may be found both in estuarine areas with juveniles and also in marine waters with adults. The scientific literature may also refer to animals in this life stage as “late-stage juveniles” or “marine migrants”. As a group, juveniles and subadults range in size from approximately 300-1500 mm TL. However, they are distinguished by differences in their occurrence within estuarine and marine waters.

Adults are the largest life stage. These are sexually mature individuals of 1500+ mm TL and 5 years of age or older. They may be found in freshwater riverine habitats on the spawning grounds or making migrations to and from the spawning grounds. They also use estuarine waters seasonally, principally in the spring through fall and will range widely in marine waters during the winter. After emigration from the natal estuary, subadults and adults travel within the marine environment, typically in waters less than 50 m in depth, using coastal bays, sounds, and ocean waters (Collins and Smith 1997; Dovel and Berggren 1983; Dunton et al. 2010; Erickson et al. 2011; Laney et al. 2007; Murawski et al. 1977; Savoy and Pacileo 2003; Smith 1985; Stein et al. 2004b; Vladykov and Greely 1963a; Welsh et al. 2002; Wirgin and King 2011).

Atlantic sturgeon populations show clinal variation, with a general trend of faster growth and earlier age at maturity in more southern systems. Atlantic sturgeon mature between the ages of 5 and 19 years in South Carolina (Smith et al. 1982), between 11 and 21 years in the Hudson River (Young et al. 1988), and between 22 and 34 years in the St. Lawrence River (Scott and Crossman 1973b). Atlantic sturgeon likely do not spawn every year. Multiple studies have shown that spawning intervals range from 1 to 5 years for males (Caron et al. 2002; Collins et al. 2000b; Smith 1985) and 2 to 5 years for females (Stevenson and Secor 1999; Van Eenennaam et al. 1996; Vladykov and Greely 1963b). Fecundity (number of eggs) of Atlantic sturgeon has been correlated with age and body size, with egg production ranging from 400,000 to 8,000,000 eggs per female per year (Dadswell 2006; Smith et al. 1982; Van Eenennaam and Doroshov 1998). The average age at which 50% of maximum lifetime egg production is achieved is estimated to be 29 years, approximately 3 to 10 times longer than for other bony fish species examined (Boreman 1997b).

Spawning adult Atlantic sturgeon generally migrate upriver in spring to early summer, which occurs in February-March in southern systems, April-May in Mid-Atlantic systems, and May-July in Canadian systems (Bain 1997; Caron et al. 2002; Murawski et al. 1977; Smith 1985; Smith and Clugston 1997). In the Carolina DPS, Smith et al. (2015) confirmed a fall spawning run in the Roanoke River, North Carolina; however, they report a spring spawning run is also likely occurring. Fall spawning runs have also been confirmed in the Edisto and Altamaha rivers, in the South Atlantic DPS. This suggests that a fall spawn is occurring in a number of southern rivers (Collins et al. 2000b; Ingram and Peterson 2016; McCord et al. 2007; Moser et al. 1998; Rogers and Weber 1995; Weber and Jennings 1996). Fall spawning periods tend to be late summer to late fall (August – November) (Smith et al. 2015)(Collins et al. 2000b; Ingram and Peterson 2016; McCord et al. 2007; Moser et al. 1998; Rogers and Weber 1995; Weber and Jennings 1996). 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 (cm) per

second and depths are 3-27 meters (m) (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973b). Males commence upstream migration to the spawning sites when waters reach around 6°C (Dovel and Berggren 1983; Smith 1985; Smith et al. 1982) with females following a few weeks later when water temperatures are closer to 12° or 13°C (Collins et al. 2000a; Dovel and Berggren 1983; Smith 1985). Atlantic sturgeon spawning occurs over hard substrate, such as cobble, gravel, or boulders, which the highly adhesive sturgeon eggs adhere to (Gilbert 1989; Smith and Clugston 1997).

Hatching occurs approximately 94-140 hours after egg deposition and larvae assume a demersal existence (Smith et al. 1980). The yolk sac larval stage is completed in about 8-12 days, during which time the larvae move downstream to rearing grounds (Kynard and Horgan 2002). During the first half of their migration downstream, movement is limited to night. During the day, larvae use benthic structure (e.g., gravel and rocks) as refugia (Kynard and Horgan 2002). During the latter half of migration when larvae are more fully developed, movement to rearing grounds occurs both day and night. Salinities of 5-10 ppt are known to cause mortality at this young stage (Bain 1997; Cech and Doroshov 2005; Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into brackish waters, and eventually become residents in estuarine waters for months or years.

During their first 2 years, juvenile Atlantic sturgeon remain in the estuaries of their natal rivers, which may include both fresh and brackish channel habitats below the head of tide (Hatin et al. 2007). Estuarine habitats are important for juveniles, serving as nursery areas by providing abundant foraging opportunities, as well as thermal and salinity refuges, for facilitating rapid growth. Some juveniles will take up residency in non-natal rivers that lack active spawning sites (Bain 1997). Residency time of young Atlantic sturgeon in estuarine areas varies between one and six years (Schueller and Peterson 2010; Smith 1985), after which Atlantic sturgeon start their outward migration to the marine environment. However, by age 5, most juveniles have likely completed their transition to saltwater and become marine migratory juveniles (i.e., subadults) that are frequently encountered in estuaries of non-natal river (Bahr and Peterson 2016). Once salt tolerant, migration from the estuaries to the sea is cued by water temperature and velocity. Adult Atlantic sturgeon will reside in the marine habitat during the non-spawning season and forage extensively. Coastal migrations by adult Atlantic sturgeon are extensive and are known to occur over sand and gravel substrate (Greene et al. 2009). Atlantic sturgeon remain in the marine habitat until the waters begin to warm, at which time ripening adults migrate back to their natal rivers to spawn.

Status and Population Dynamics

At the time Atlantic sturgeon were listed, the best available abundance information for each of the 5 DPSs was the estimated number of adult Atlantic sturgeon spawning in each of the rivers on an annual basis. The estimated number of annually spawning adults in each of the river subpopulations is insufficient to quantify the total population numbers for each DPS of Atlantic sturgeon due to the lack of other necessary accompanying life history data. An attempt to estimate total ocean population numbers of adults and subadults was completed in 2012 using data from the Northeast Area Monitoring and Assessment Program (NEAMAP). NEAMAP trawl surveys were conducted from Cape Cod,

Massachusetts, to Cape Hatteras, North Carolina, in nearshore waters to depths of 60 ft from fall 2007 through spring 2012. The results of these surveys, assuming 50% gear efficiency (i.e., assumption that the gear will capture some, but not all, of the sturgeon in the water column along the tow path, and the survey area is only a portion of Atlantic sturgeon habitat), are presented in Table 3.7. It is important to note that the NEAMAP surveys were conducted primarily in the Northeast and may underestimate the actual population abundances of the Carolina and South Atlantic DPSs, which are likely more concentrated in the Southeast since they originated from and spawn there. However, the total ocean population abundance estimates listed in Table 3.7 currently represent the best available population abundance estimates for the 5 U.S. Atlantic sturgeon DPSs.

Table 3.7. Summary of Calculated Population Estimates based upon the NEAMAP Survey Swept Area, Assuming 50% Efficiency (NMFS 2013)

DPS	Estimated Ocean Population Abundance	Estimated Ocean Population of Adults	Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries)
South Atlantic	14,911	3,728	11,183
Carolina	1,356	339	1,017
Chesapeake Bay	8,811	2,203	6,608
New York Bight	34,566	8,642	25,925
Gulf of Maine	7,455	1,864	5,591

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 River (ACE) Basins southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, Florida. The marine range of Atlantic sturgeon from the South Atlantic DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida.

Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in Georgia prior to 1890.

The South Atlantic DPS historically likely supported 8 spawning subpopulations. At the time of listing in 2012, only 6 spawning subpopulations were believed to have existed: the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers. The two remaining spawning subpopulations in the Broad-Coosawatchie River and St. Marys River were believed to be extinct. However, new information provided from the capture of juvenile Atlantic sturgeon suggests the spawning subpopulation in the St. Marys River is not extinct and continues to exist, albeit at very low levels. Two of the spawning subpopulations in the South Atlantic DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. These two spawning subpopulations are likely less than 6% of their historic abundance. The abundance of the remaining 3 spawning subpopulations in the South Atlantic DPS is likely less than 1% of their historical abundance (ASSRT 2007). There are an estimated 343 adults that spawn annually in the Altamaha River and less than 300 adults spawning annually (total of both sexes) in the river systems where spawning still occurs (75 FR 61904, October 6, 2010). 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.

In 2017, the Atlantic States Marine Fisheries Commission (ASMFC) completed an Atlantic Sturgeon Benchmark Stock Assessment ("Assessment") (ASMFC 2017). The purpose of the assessment was to evaluate the status of Atlantic sturgeon along the U.S. Atlantic coast (ASMFC 2017). The assessment considered the status of each DPS individually, as well as all 5 DPSs collectively as a single unit. The assessment determined the abundance of the South Atlantic DPS is "depleted" relative to historical levels. The assessment concluded there was not enough information available to assess the abundance of the DPS since the implementation of the 1998 fishing moratorium. However, it did conclude there was 40% probability the South Atlantic DPS is still subjected to mortality levels higher than those determined acceptable in the 2017 assessment.

The Assessment also estimated effective population sizes (N_e) when possible. Effective population size is generally considered to be the number of individuals that contribute offspring to the next generation. More specifically, based on genetic differences between animals in a given year, or over a given period of time, researchers can estimate the number of adults needed to produce that level of genetic diversity. Generally, a minimum N_e of 100 individuals is considered the threshold required to limit the loss in total fitness from in-breeding depression to <10%; while an N_e greater than 1,000 is the recommended minimum to maintain evolutionary potential (ASMFC 2017; Frankham et al. 2014). N_e is useful for defining abundance levels where populations are at risk of loss of genetic fitness (ASMFC 2017). For the South Atlantic DPS, the assessment reported an N_e for the Savannah, Ogeechee, Altamaha rivers, and Edisto. In the Savannah River, samples from 98 individuals collected from 2000-2013 produced an estimated N_e of 126.5 individuals. In the Ogeechee River, 115 samples were collected from 2003-2015 and produced an estimated N_e of 32.2 individuals. The sample size from the Altamaha River was the largest ($n=186$), collected from 2005-2015, and produced an estimated N_e of 111.9 individuals (ASMFC 2017). For the Edisto River, samples collected from 109 individuals from 1996-2005, produced an estimated N_e of 55.4 individuals. Farrae et al. (2017) also estimated an N_e of 48.0 individuals for fall spawning fish in the Edisto River. While not inclusive of all

the spawning rivers in the South Atlantic DPS, these estimates suggest there is a risk for inbreeding depression ($N_e < 100$) in two (Edisto and Ogeechee rivers) of those four rivers, and loss of evolutionary potential ($N_e < 1000$) in all four. This information suggests there at least some inbreeding depression within the DPS and loss of evolutionary potential throughout all of it. The NEAMAP model estimates a minimum ocean population for the entire South Atlantic DPS of 14,911 Atlantic sturgeon, of which 3,728 are adults.

Carolina DPS

The Carolina DPS includes all Atlantic sturgeon that are spawned in the watersheds (including all rivers and tributaries) from the 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 South Atlantic DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida.

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. Currently, there are believed to be only 5 of 7-10 historical spawning subpopulations remaining in the Carolina DPS. These populations include the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Yadkin-Pee Dee River populations. There may also be spawning subpopulations in the Neuse, Santee, and Cooper Rivers, though it is uncertain. The abundances of the other 5 river spawning subpopulations within the DPS are estimated to have fewer than 300 spawning adults, or less than 3% of what they were historically (ASSRT 2007). We determined spawning was occurring if YOY were observed, or mature adults were present, in freshwater portions of a system. 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 Sampit and Ashley Rivers in South Carolina were documented to have spawning subpopulations at one time. Yet, the spawning subpopulation in the Sampit River is believed to be extirpated and the current status of the spawning subpopulation in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning subpopulations.

The Assessment determined the Carolina DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (67%) that the Carolina DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a relatively high probability (75%) the Carolina DPS is still subjected to mortality levels higher than determined acceptable in the 2017 assessment (ASMFC 2017).

For the Carolina DPS, the Assessment only reported N_e for the Albemarle Sound. Based on samples collected from 37 individuals from 1998-2008, the Assessment estimated an N_e of 14.2 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the Carolina DPS, this estimate suggests there is a risk for both inbreeding depression ($N_e < 100$) and loss of evolutionary potential ($N_e < 1000$) in the DPS, assuming Albemarle Sound

is representative of the entire DPS. The NEAMAP model estimates a minimum ocean population for the entire Carolina DPS of 1,356 Atlantic sturgeon, of which 339 are adults.

Chesapeake Bay DPS

The Chesapeake Bay DPS is comprised of Atlantic sturgeon that originate from rivers that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, Virginia. The marine range of Atlantic sturgeon from the Chesapeake Bay DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida.

Historically, the Chesapeake Bay DPS likely supported more than 10,000 spawning adults (ASSRT 2007; KRRMP 1993; Secor 2002). Currently, there are 4 known spawning subpopulations for the Chesapeake Bay DPS, one each for the Pamunkey River and for Marshyhope Creek, and two for the James River (Balazik et al. 2017; Balazik et al. 2012a) (Balazik and Musick 2015; Hager et al. 2014; Richardson and Secor 2016; Richardson and Secor 2017). Atlantic sturgeon that are spawned elsewhere are known to use waters of the Chesapeake Bay for other life functions, such as foraging and as juvenile nursery habitat, before entering the marine system as subadults (ASSRT 2007; Grunwald et al. 2008; Vladykov and Greely 1963b; Wirgin et al. 2007).

The existence of the Pamunkey River spawning subpopulation was identified in 2013 after the capture of spawning condition adults (e.g., males expressing milt, and females with eggs) within tidal freshwater of the river during the late summer to early fall (i.e., August - October) (Hager et al. 2014). Based on the capture of 17 sturgeon, Kahn et al. (2014) estimated 75 adults (95% confidence interval = 17–168 adults) spawned in the river in 2013. There are no other estimates of abundance for this spawning subpopulation or trends in abundance.

The Marshyhope Creek spawning subpopulation was identified in 2014, likewise after the capture of spawning condition adults during the late summer to early fall. Twenty-six adults, including males expressing milt and females with ripe eggs, have been captured in Marshyhope Creek since 2014. DNA analysis is ongoing to determine whether the sturgeon are part of a naturally occurring population or are hatchery fish that were released into the Nanticoke River in 1996 (Richardson and Secor 2016; Richardson and Secor 2017; Secor et al. 2000). There are no estimates of abundance or trends in abundance for this spawning subpopulation.

At the time of listing, the James River was the only known spawning river for the Chesapeake Bay DPS and spawning was believed to occur only in the spring, from approximately April –May, based on historical and current evidence (ASSRT 2007). Subsequently, new information for when and where spawning-condition adults were captured and tracked in the river led to the conclusion that Atlantic sturgeon spawn in the James River in both the spring and in the late summer to early fall (Balazik et al. 2012a; Balazik and Musick 2015). The results of genetic analyses support that the adults are two separate spawning groups. The genetic analyses also informed the effective population size of each group which were similar (Fall: $N_e = 46$ (95% CI: 32 ± 71), Spring: $N_e = 44$ (95% CI: 26 ± 79)) despite differences in the number of adults captured from each spawning

subpopulation. From 2007 to 2016, 507 individual fall run Atlantic sturgeon were captured during the fall spawning and 40 individual Atlantic sturgeon were captured during the spring spawning (Balazik et al. 2017). This is a minimum count of the number of adult Atlantic sturgeon in the James River during the time period because capture efforts did not occur in all areas and at all times when Atlantic sturgeon were present in the river. There are no other estimates of abundance or trends in abundance for the James River spawning subpopulations.

The 2017 Assessment determined the Chesapeake Bay DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively low probability (37%) that the Chesapeake Bay DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 30% probability the Chesapeake Bay DPS is still subjected to mortality levels higher than determined acceptable in the 2017 assessment.

The 2017 Assessment reported N_e for the York and James rivers in the Chesapeake Bay DPS. In the York River, samples from 136 individuals collected from 2013-2015 produced an estimated N_e of 7.8 individuals (ASMFC 2017). In the James River, 346 samples were collected from 1998-2015 and produced an estimated N_e of 40.9 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the Chesapeake Bay DPS, these estimates at least hint that there is a risk for both inbreeding depression ($N_e < 100$) and loss of evolutionary potential ($N_e < 1000$) in the DPS. The NEAMAP model estimates a minimum ocean population for the entire DPS of 8,811 Atlantic sturgeon, of which 2,319 are adults.

New York Bight DPS

The New York Bight DPS includes all anadromous Atlantic sturgeon that spawn in the watersheds that drain into coastal waters from Chatham, Massachusetts, to the Delaware-Maryland border on Fenwick Island. The marine range of Atlantic sturgeon from the New York Bight DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida.

Within this range, Atlantic sturgeon historically spawned in the Connecticut, Delaware, Hudson, and Taunton Rivers (ASSRT 2007; Murawski et al. 1977; Secor 2002). Spawning still occurs in the Delaware and Hudson Rivers, and evidence of spawning was recently documented in the Connecticut River (ASSRT 2007; Savoy et al. 2017). Atlantic sturgeon that are spawned elsewhere continue to use habitats within the Connecticut and Taunton Rivers for other life functions (ASSRT 2007; Savoy 2007; Wirgin and King 2011)

Prior to the onset of expanded fisheries exploitation of sturgeon in the 1800s, a conservative historical estimate for the Hudson River Atlantic sturgeon population was 10,000 adult females (Secor 2002). Current population abundance is likely at least one order of magnitude smaller than historical levels (ASSRT 2007; Kahnle et al. 2007; Secor 2002). Based on data collected from 1985-1995, 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 (Kahnle et al. 2007). Kahnle (2007; 1998) 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. At the time of listing, available data on abundance of juvenile Atlantic sturgeon in the Hudson River Estuary indicated a substantial drop in production of young since the mid-1970s (Kahnle et al. 1998). A decline appeared to occur in the mid- to late-1970s followed by a secondary drop in the late 1980s (ASMFC 2010; Kahnle et al. 1998; Sweka et al. 2007). Catch-per-unit-effort (CPUE) data suggest that recruitment has remained depressed relative to catches of juvenile Atlantic sturgeon in the estuary during the mid- to late 1980s (ASMFC 2010; Sweka et al. 2007). From 1985-2007, there were significant fluctuations in CPUE. The number of juveniles appears to have declined between the late 1980s and early 1990s. While the CPUE is generally higher in the 2000s as compared to the 1990s, significant annual fluctuations make it difficult to discern any trend. The CPUEs from 2000-2007 are generally higher than those from 1990-1999; however, they remain lower than the CPUEs observed in the late 1980s. Standardized mean catch per net set from the NYSDEC juvenile Atlantic sturgeon survey have had a general increasing trend from 2006 – 2015, with the exception of a dip in 2013. There is currently not enough information regarding any life stage to establish a trend for the Hudson River population (ASMFC 2010; Sweka et al. 2007).

There is no abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800s indicate that this was historically a large population, with an estimated 180,000 adult females prior to 1890 (Secor 2002; Secor and Waldman 1999). Fisher (2009) sampled the Delaware River in 2009 to target YOY Atlantic sturgeon. The effort captured 34 YOY. Brundage and O'Herron (2003) also collected 32 YOY Atlantic sturgeon from the Delaware River in a separate study. Fisher (2011) reports that genetics information collected from 33 of the 2009 year class YOY indicates that at least 3 females successfully contributed to the 2009 year class. The capture of YOY in some years since 2009 shows that successful spawning is still occurring in the Delaware River. Based on the capture of juvenile Atlantic sturgeon in the Delaware River, researchers estimated there were 3,656 (95% CI = 1,935–33,041) age 0-1 juvenile Atlantic sturgeon in the Delaware River subpopulation in 2014 (Hale et al. 2016). However, the relatively low numbers of captured adults suggest the existing riverine subpopulation is limited in size. For example, of the 261 adult-sized Atlantic sturgeon captured for scientific purposes off the Delaware Coast between 2009 and 2012, 100 were subsequently identified by genetics analysis to belong to the Hudson River subpopulation while only 36 belonged to the Delaware River subpopulation (Wirgin et al. 2015). Similar to the Hudson River, there is currently not enough information to determine a trend for the Delaware River population. The ASSRT (2007) suggested that there may be less than 300 spawning adults per year for the Delaware River portion of the New York Bight DPS.

The 2017 Assessment determined the New York Bight DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (75%) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 31% probability the New York Bight DPS is still subjected to mortality levels higher than determined acceptable in the 2017 assessment (ASMFC 2017).

The 2017 Assessment reported N_e for the Hudson and Delaware rivers in the New York Bight DPS. In the Hudson River, samples from 337 individuals collected from 1996-2015 produced an estimated N_e of 144.2 individuals (ASMFC 2017). In the Delaware River, 181 samples were collected from 2009-2015 and produced an estimated N_e of 56.7 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the New York Bight DPS, the estimates for the Hudson River suggests that spawning subpopulation may be large enough to avoid inbreeding depression ($N_e < 100$); however, the Delaware River spawning subpopulation may still be at risk. Both spawning subpopulations are likely at risk losing evolutionary potential ($N_e < 1000$). The NEAMAP model estimates a minimum ocean population for the entire DPS of 34,566 Atlantic sturgeon, of which 8,642 are adults.

Gulf of Maine DPS

The Gulf of Maine DPS includes all anadromous Atlantic sturgeons that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, Massachusetts. The marine range of Atlantic sturgeon from the Gulf of Maine DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida.

Within this range, Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (ASSRT 2007). Spawning still occurs in the Kennebec River, and captures of adult Atlantic sturgeon in the Androscoggin River, including a ripe male, over suitable spawning grounds during the spawning season confirm likely spawning; however, Atlantic sturgeon eggs and larvae have not yet been recovered in the Androscoggin (Wippelhauser pers. comm. 2018). The movement of subadult and adult sturgeon between rivers, including to and from the Kennebec River and the Penobscot River, demonstrates that coastal and marine migrations are key elements of Atlantic sturgeon life history for the Gulf of Maine DPS as well as likely throughout the entire range (ASSRT 2007; Fernandes et al. 2010).

Historically, the Gulf of Maine DPS likely supported more than 10,000 spawning adults (ASSRT 2007; KRRMP 1993; Secor 2002). Other than the NEAMAP based estimates presented above, there are no empirical abundance estimates for the Gulf of Maine DPS. The ASSRT (2007) presumed that the Gulf of Maine DPS was comprised of less than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over two time periods, 1977-1981 and 1998-2000, resulted in the capture of 9 adult Atlantic sturgeon (Squiers 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the gear used may not have been selective for larger, adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies.

The 2017 Assessment determined the Gulf of Maine DPS abundance is "depleted" relative to historical levels. It also determined there is a 51% probability Gulf of Maine DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 74% probability the Gulf of Maine DPS is still subjected to mortality levels higher than determined acceptable in the 2017 assessment (ASMFC 2017).

The 2017 Assessment reported an N_e for the St. Lawrence, St. John, and Kennebec rivers in the Gulf of Maine DPS. In the St. Lawrence, samples from 30 individuals collected in 2013 produced an estimated N_e of 39.0 individuals (ASMFC 2017). In the St. John River, 31 samples were collected from 1991-1993 and produced an estimated N_e of 115.0 individuals (ASMFC 2017). For the Kennebec River, samples from 52 individuals were collected from 1980-2011, and produced an estimated N_e of 63.4 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the Gulf of Maine DPS, the effective population size estimate for the St. John River suggests that spawning subpopulation may be large enough to avoid inbreeding depression ($N_e < 100$); however, the estimates for the remaining two rivers suggests those spawning subpopulations may be at risk. All three spawning subpopulations are likely at risk losing evolutionary potential ($N_e < 1000$). The NEAMAP model estimates a minimum ocean population for the entire DPS of 7,455 Atlantic sturgeon, of which 1,864 are adults.

Viability of Atlantic Sturgeon DPSs

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 5 DPSs on the East Coast put them in danger of extinction throughout their range. None of the riverine spawning subpopulations are large or stable enough to provide with any level of certainty for continued existence of any of the DPSs. Although the largest impact that caused the precipitous decline of the species has been prohibited (directed fishing), the Atlantic sturgeon population sizes within each DPS have remained relatively constant at greatly reduced levels for 100 years. The largest Atlantic sturgeon population in the United States, the Hudson River population within the New York Bight DPS, is estimated to have only 870 spawning adults each year. The Altamaha River population within the South Atlantic DPS is the largest Atlantic sturgeon population in the Southeast and only has an estimated 343 adults spawning annually. All other Atlantic sturgeon river populations in the U.S. are estimated to have less than 300 spawning adults annually.

Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry 1971; Shaffer 1981; Soulé 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life span allows multiple opportunities to contribute to future generations, it also increases the time frame over which exposure to the multitude of threats facing Atlantic sturgeon can occur.

The viability of the Atlantic sturgeon DPSs depends on having multiple self-sustaining riverine spawning subpopulations and maintaining suitable habitat to support the various life functions (spawning, feeding, growth) of Atlantic sturgeon populations. Because a DPS is a group of populations, the stability, viability, and persistence of individual populations affects the persistence and viability of the larger DPS. The loss of any population within a DPS will result in: (1) a long-term gap in the range of the DPS that is unlikely to be recolonized; (2) loss of reproducing individuals; (3) loss of genetic

biodiversity; (4) potential loss of unique haplotypes; (5) potential loss of adaptive traits; (6) reduction in total number; and (7) potential for loss of population source of recruits. The loss of a population will negatively impact the persistence and viability of the DPS as a whole, as fewer than 2 individuals per generation spawn outside their natal rivers (King et al. 2001; Waldman et al. 2002; Wirgin et al. 2000). The persistence of individual populations, and in turn the DPS, depends on successful spawning and rearing within the freshwater habitat, the immigration into marine habitats to grow, and then the return of adults to natal rivers to spawn.

Threats

Atlantic sturgeon were once numerous along the East Coast until fisheries for their meat and caviar reduced the populations by over 90% in the late 1800s. Fishing for Atlantic sturgeon became illegal in state waters in 1998 and in remaining U.S. waters in 1999. Dams, dredging, poor water quality, and accidental catch (bycatch) by fishers continue to threaten Atlantic sturgeon. Though Atlantic sturgeon populations appear to be increasing in some rivers, other river populations along the East Coast continue to struggle and some have been eliminated entirely. The 5 DPSs of Atlantic sturgeon were listed as threatened or endangered under the ESA primarily as a result of a combination of habitat restriction and modification, overutilization (i.e., being taken as bycatch) in commercial fisheries, and the inadequacy of regulatory mechanisms in ameliorating these impacts and threats.

Dams

Dams for hydropower generation, flood control, and navigation adversely affect Atlantic sturgeon by impeding access to spawning, developmental, and foraging habitat, modifying free-flowing rivers to reservoirs, physically damaging fish on upstream and downstream migrations, and altering water quality in the remaining downstream portions of spawning and nursery habitat (ASSRT 2007). Attempts to minimize the impacts of dams using measures such as fish passage have not proven beneficial to Atlantic sturgeon, as they do not regularly use existing fish passage devices, which are generally designed to pass pelagic fish (i.e., those living in the water column) rather than bottom-dwelling species, like sturgeon. However, NMFS continues to evaluate ways to effectively pass sturgeon above and below man-made barriers. For example, large nature-like fishways (e.g., rock ramps) hold promise as a mechanism for successful passage.

Within the range of the Carolina DPS, dams have restricted Atlantic sturgeon spawning and juvenile developmental habitat by blocking over 60% of the historical sturgeon habitat upstream of the dams in the Cape Fear and Santee-Cooper River systems. Water quality (velocity, temperature, and DO downstream of these dams, as well as on the Roanoke River, has been reduced, which modifies and restricts the extent of spawning and nursery habitat for the Carolina DPS.

Within in the range of the South Atlantic DPS, on the Savannah River, the New Savannah Bluff Lock and Dam (NSBL&D) at the City of Augusta, is located just a few kilometers below impassible rapids, denying Atlantic sturgeon access to 7% of its historically available habitat (ASSRT 1998). However, the Augusta Shoals, the only rocky shoal habitat on the Savannah River and the former primary spawning habitat for Atlantic sturgeon in the river (Duncan et al. 2003; Marcy et al. 2005; USFWS 2003; Wrona et al. 2007), is located above NSBL&D, and is currently inaccessible to Atlantic sturgeon. So,

while Atlantic sturgeon have access to the majority of historical habitat in terms of unimpeded river miles, only a small amount of spawning habitat exists downstream of the NSBL&D and the vast majority of the rocky freshwater spawning habitat they need is inaccessible as a result of the NSBL&D.

Within the range of the New York Bight DPS, the Holyoke Dam on the Connecticut River blocks further upstream passage; however, the extent that Atlantic sturgeon historically would have used habitat upstream of Holyoke is unknown. Connectivity may be disrupted by the presence of dams on several smaller rivers in the New York Bight region. Connectivity is disrupted by the presence of dams on several rivers in the range of the Gulf of Maine DPS. Within the Gulf of Maine DPS, access to historical spawning habitat is most severely impacted in the Merrimack River (ASSRT 2007). Construction of the Essex Dam blocked the migration of Atlantic sturgeon to 58% of its historically available habitat (ASSRT 2007). The extent that Atlantic sturgeon are affected by operations of dams in the Gulf of Maine region is currently unknown

Dredging

Riverine, nearshore, and offshore areas are often dredged to support commercial shipping and recreational boating, construction of infrastructure, and marine mining. Environmental impacts of dredging include the direct removal/burial of prey species; turbidity/siltation effects; contaminant resuspension; noise/disturbance; alterations to hydrodynamic regime and physical habitat; and actual loss of riparian habitat (Chytalo 1996; Winger et al. 2000). According to Smith and Clugston (1997), dredging and filling impact important habitat features of Atlantic sturgeon as they disturb benthic fauna, eliminate deep holes, and alter rock substrates.

In the South Atlantic DPS, maintenance dredging is currently modifying Atlantic sturgeon nursery habitat in the Savannah River. Modeling indicates that the proposed deepening of the navigation channel will result in reduced DO and upriver movement of the salt wedge, restricting spawning habitat. Dredging is also modifying nursery and foraging habitat in the St. Johns River. For the Carolina DPS, dredging in spawning and nursery grounds modifies the quality of the habitat and is further restricting the extent of available habitat in the Cape Fear and Cooper Rivers, where Atlantic sturgeon habitat has already been modified and restricted by the presence of dams. Dredging for navigational purposes is suspected of having reduced available spawning habitat for the Chesapeake Bay DPS in the James River (ASSRT 2007; Bushnoe et al. 2005; Holton and Walsh 1995). 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. Many rivers in the range of the Gulf of Maine DPS, including the Kennebec River, also have navigation channels that are maintained by dredging. Dredging outside of federal channels and in-water construction occurs throughout the range of the Chesapeake Bay, New York Bight and Gulf of Maine DPSs.

Water Quality

Atlantic sturgeon rely on a variety of water quality parameters to successfully carry out their life functions. Low DO and the presence of contaminants modify the quality of Atlantic sturgeon habitat and in some cases, restrict the extent of suitable habitat for life

functions. Secor (1995) noted a correlation between low abundances of sturgeon during this century and decreasing water quality caused by increased nutrient loading and increased spatial and temporal frequency of hypoxic (low oxygen) conditions. Of particular concern is the high occurrence of low DO coupled with high temperatures in the river systems throughout the range of the Carolina and South Atlantic DPSs in the Southeast. Sturgeon are more highly sensitive to low DO than other fish species (Niklitschek and Secor 2009a; Niklitschek and Secor 2009b) and low DO in combination with high temperature is particularly problematic for Atlantic sturgeon. Studies have shown that juvenile Atlantic sturgeon experience lethal and sublethal (metabolic, growth, feeding) effects as DO drops and temperatures rise (Niklitschek and Secor 2005; Niklitschek and Secor 2009a; Niklitschek and Secor 2009b; Secor and Gunderson 1998).

Reductions in water quality from terrestrial activities 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. In the Pamlico and Neuse systems of the Carolina DPS, nutrient-loading and seasonal anoxia are occurring, associated in part with concentrated animal feeding operations (CAFOs). Heavy industrial development and CAFOs have degraded water quality in the Cape Fear River. Water quality in the Waccamaw and Yadkin-Pee Dee Rivers has been affected by industrialization and riverine sediment samples contain high levels of various toxins, including dioxins.

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 (ASMFC 1998; ASSRT 2007; Pyzik et al. 2004). These conditions contribute to reductions in DO levels throughout the bay. The availability of nursery habitat, in particular, may be limited given the recurrent hypoxia (low DO) conditions within the Bay (Niklitschek and Secor 2005; Niklitschek and Secor 2010).

Both the Hudson and Delaware Rivers, as well as other rivers in the New York Bight region, were heavily polluted in the past from industrial and sewer discharges. In the past, many rivers in Maine, including the Androscoggin River, were heavily polluted from industrial discharges from pulp and paper mills. While water quality has improved and most discharges are limited through regulations, many pollutants persist in the benthic environment of the New York Bight and Gulf of Maine DPSs. It is particularly problematic if pollutants are present on spawning and nursery grounds, as developing eggs and larvae are particularly susceptible to exposure to contaminants.

Atlantic sturgeon may also be particularly susceptible to impacts from environmental contamination because they are long-lived, benthic feeders. Sturgeon feeding in estuarine habitats near urbanized areas may be exposed to numerous suites of contaminants within the substrate. Contaminants, including toxic metals, polychlorinated aromatic hydrocarbons (PAHs), organophosphate and organochlorine pesticides, polychlorinated biphenyls (PCBs), and other chlorinated hydrocarbon compounds can have substantial

deleterious effects on aquatic life. These elements and compounds can cause acute lesions, growth retardation, and reproductive impairment in fishes (ASSRT 2007; Cooper 1989; Sindermann 1994). Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but their long-term effects are not known (Ruelle and Henry 1992; Ruelle and Keenlyne 1993). Elevated levels of contaminants, including chlorinated hydrocarbons, in several other fish species are associated with reproductive impairment (Cameron et al. 1992; Drevnick and Sandheinrich 2003; Hammerschmidt et al. 2002; Longwell et al. 1992), reduced egg viability (Billsson et al. 1998; Giesy et al. 1986; Mac and Edsall 1991; Matta et al. 1997; Von Westernhagen et al. 1981a), reduced survival of larval fish (Berlin et al. 1981; Giesy et al. 1986), delayed maturity (Jorgensen (Jorgensen et al. 2004b) and posterior malformations (Billsson et al. 1998). Pesticide exposure in fish may affect antipredator and homing behavior, reproductive function, physiological development, and swimming speed and distance (Beauvais et al. 2000; Moore and Waring 2001; Scholz et al. 2000; Waring and Moore 2004). It should be noted that the effect of multiple contaminants or mixtures of compounds at sub-lethal levels on fish has not been adequately studied. Atlantic sturgeon use marine, estuarine, and freshwater habitats and are in direct contact through water, diet, or dermal exposure with multiple contaminants throughout their range (ASSRT 2007). Trace metals, trace elements, or inorganic contaminants (mercury, cadmium, selenium, lead, etc.) are another suite of contaminants occurring in fish. Post (1987) states that toxic metals may cause death or sub-lethal effects to fish in a variety of ways and that chronic toxicity of some metals may lead to the loss of reproductive capabilities, body malformation, inability to avoid predation, and susceptibility to infectious organisms.

Water Quantity

Water allocation issues are a growing threat in the Southeast and exacerbate existing water quality problems. Taking water from one basin and transferring it to another fundamentally and irreversibly alters natural water flows in both the originating and receiving basins, which can affect DO levels, temperature, and the ability of the basin of origin to assimilate pollutants (GWC 2006). Water quality within the river systems in the range of the South Atlantic and Carolina DPSs is negatively affected by large water withdrawals. Known water withdrawals of over 240 million gallons per day (mgd) are permitted from the Savannah River for power generation and municipal uses. However, permits for users withdrawing less than 100,000 gallons per day are not required, so actual water withdrawals from the Savannah and other rivers within the range of the South Atlantic DPS are likely much higher.

In the range of the Carolina DPS, 20 interbasin water transfers in existence prior to 1993, averaging 66.5 mgd, were authorized at their maximum levels without being subjected to an evaluation for certification by the North Carolina Department of Environment and Natural Resources or other resource agencies. Since the 1993 legislation requiring certificates for transfers, almost 170 mgd of interbasin water withdrawals have been authorized, with an additional 60 mgd, pending certification. The removal of large amounts of water from these systems will alter flows, temperature, and DO. Water shortages and “water wars” are already occurring in the rivers occupied by the South Atlantic and Carolina DPSs and will likely be compounded in the future by population growth and potentially by climate change.

Climate Change

The Intergovernmental Panel on Climate Change (IPCC) projects with high confidence that higher water temperatures and changes in extremes, including floods and droughts, will affect water quality and exacerbate many forms of water pollution – from sediments, nutrients, dissolved organic carbon, pathogens, pesticides, and salt, as well as thermal pollution – with possible negative impacts on ecosystems (IPCC 2008). In addition, sea level rise is projected to extend areas of salinization of groundwater and estuaries, resulting in a decrease of freshwater availability for humans and ecosystems in coastal areas. Some of the most heavily populated areas are low-lying, and the threat of salt water entering into its aquifers with projected sea level rise is a concern (USGRG 2004). Existing water allocation issues would be exacerbated, leading to an increase in reliance on interbasin water transfers to meet municipal water needs, further stressing water quality.

Dams, dredging, and poor water quality have already modified and restricted the extent of suitable habitat for Atlantic sturgeon spawning and nursery habitat. Changes in water availability (depth and velocities) and water quality (temperature, salinity, DO, contaminants, etc.) in rivers and coastal waters inhabited by Atlantic sturgeon resulting from climate change will further modify and restrict the extent of suitable habitat for Atlantic sturgeon. Effects could be especially harmful since these populations have already been reduced to low numbers, potentially limiting their capacity for adaptation to changing environmental conditions (Belovsky 1987; Salwasser et al. 1984; Soulé 1987; Thomas 1990).

The effects of changes in water quality (temperature, salinity, DO, contaminants, etc.) in rivers and coastal waters inhabited by Atlantic sturgeon are expected to be more severe for those populations that occur at the southern extreme of the Atlantic sturgeon's range, and in areas that are already subject to poor water quality as a result of eutrophication. The South Atlantic and Carolina DPSs are within a region the IPCC predicts will experience overall climatic drying (IPCC 2008). Atlantic sturgeon from these DPSs are already susceptible to reduced water quality resulting from various factors: inputs of nutrients; contaminants from industrial activities and non-point sources; and interbasin transfers of water. In a simulation of the effects of water temperature on available Atlantic sturgeon habitat in Chesapeake Bay, Niklitschek and Secor (2005) found that a 1°C increase of water temperature in the bay would reduce available sturgeon habitat by 65%.

Ocean temperature in the U.S. Northeast Shelf and surrounding Northwest Atlantic waters has increased faster than the global average over the last decade (Pershing et al. 2015). New projections for the U.S. Northeast Shelf and Northwest Atlantic Ocean suggest that this region will warm two to three times faster than the global average (Saba et al. 2016). A first-of-its-kind climate vulnerability assessment, conducted on 82 fish and invertebrate species in the Northeast U.S. Shelf, concluded that Atlantic sturgeon from all five DPSs were among the most vulnerable species to global climate change (Hare et al. 2016). It is very likely that the magnitude and frequency of ecosystem changes as a result of global climate change will continue to increase, possibly at an accelerated pace, in the next 50 years regardless of any reduction in greenhouse gases, due to emissions that have already occurred (NAST 2000).

There is a high confidence, 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 pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007).

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). Expected consequences of climate change for river systems could be a decrease in the amount of dissolved oxygen in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000).

Although Atlantic sturgeon have persisted for millions of years and have experienced wide variations in global climate conditions, the current rate of climate change reported and/or anticipated to occur is faster than what we can reasonably expect Atlantic sturgeon to be able to adapt to.

Vessel Strikes

Vessel strikes are a threat to the Chesapeake Bay and New York Bight DPSs. Eleven Atlantic sturgeon were reported to have been struck by vessels on the James River from 2005 through 2007. Several of these were mature individuals. From 2007–2010, researchers documented 31 carcasses of adult Atlantic sturgeon in the tidal freshwater portion of the James River, Virginia (Balazik et al. 2012b). Twenty-six of the carcasses had gashes from vessel propellers, and the remaining 5 carcasses were too decomposed to allow determination of the cause of death (Balazik et al. 2012b). The types of vessels responsible for these mortalities could not be confirmed. Most (84%) of the carcasses were found in a relatively narrow reach that has been modified to increase shipping efficiency (Balazik et al. 2012b). Using telemetry, Balazik et al. (2012b) reported that while staging (holding in an area from hours to days, with minimal upstream or downstream movements), adult male Atlantic sturgeon spent most (62%) of their time within 1 m of the river bottom. Under the assumption that Atlantic sturgeon do not modify their behavior as a result of vessel noise, Balazik et al. (2012b) hypothesized adult male Atlantic sturgeon in the James River would rarely encounter small recreational boats or tugboats with shallow drafts. Instead, Balazik et al. (2012b) concluded vessel strike mortalities are likely caused by deep-draft ocean cargo ships, with drafts that coincide with the river depths most frequently used by the animals they tracked using telemetry. Ultimately, they estimated that current monitoring in the James River documents less than one-third of vessel strike mortalities (Balazik et al. 2012b).

From 2004–2008, 29 mortalities believed to be the result of vessel strikes were documented in the Delaware River; at least 13 of these fish were large adults. The time of year when these events occurred (predominantly May through July, with 2 in August), indicate the animals were likely adults migrating through the river to the spawning grounds. Because we do not know the percent of total vessel strikes that these observed mortalities represent, we are not able to quantify the number of individuals likely killed as a result of vessel

strikes in the Chesapeake and New York Bight DPSs. Very little is known about the effects of vessel strikes on individuals from the Carolina or South Atlantic DPSs.

Bycatch Mortality

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to Atlantic sturgeon in all 5 DPSs. Atlantic sturgeon are more sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum reproductive rates, and a large percentage of egg production occurs later in life. Based on these life history traits, Boreman (1997b) 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. Currently, there are estimates of the number of Atlantic sturgeon captured and killed in sink gillnet and otter trawl fisheries authorized by Fishery Management Plans (FMPs) in the Northeast Region (Miller and Shepherd 2011). Those estimates indicate from 2006-2010, on average there were 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%, while mortality rates in otter trawl gear are generally lower, at approximately 5%. 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. Atlantic sturgeon are incidentally captured in state and federal fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (ASMFC 2007; Stein et al. 2004a). Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

3.3.9 Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS

Four of 6 identified DPSs of scalloped hammerhead shark (*Sphyrna lewini*) were listed under the ESA by NMFS effective September 2, 2014 (79 FR 38213, July 3, 2014) (Figure 3.9). The Central and Southwest Atlantic and the Indo-West Pacific DPSs were listed as threatened, while the Eastern Atlantic and Eastern Pacific DPSs were listed as endangered. The Central and Southwest Atlantic DPS is bounded to the north by 28°N latitude, to the east by 30°W longitude, and to the south by 36°S latitude. All waters of the Caribbean Sea are within this DPS boundary, including The Bahamas' EEZ off the coast of Florida, the U.S. EEZ off Puerto Rico and the U.S. Virgin Islands, and Cuba's EEZ.

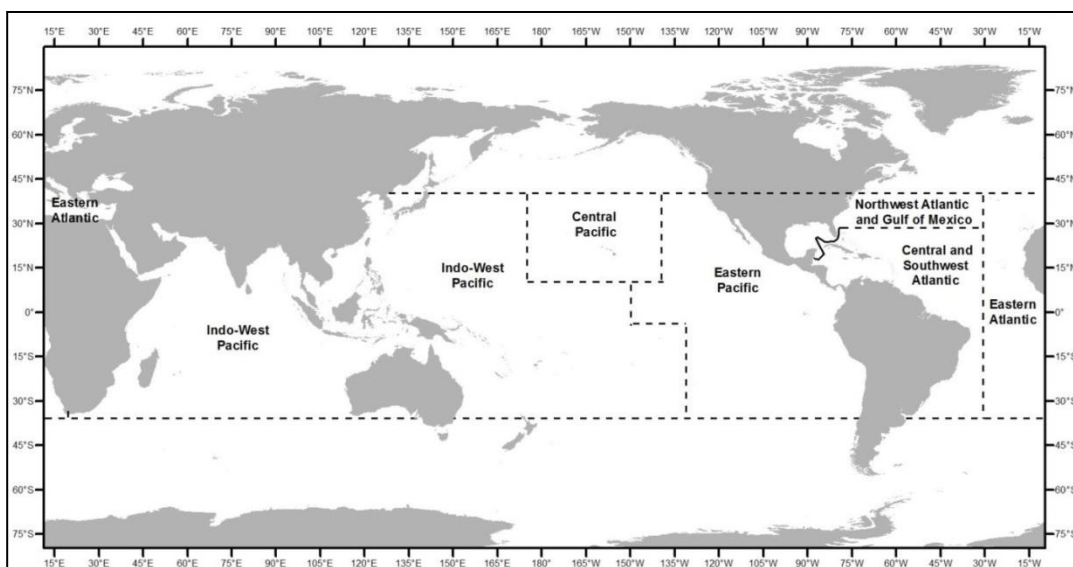


Figure 3.9 Scalloped hammerhead shark DPS boundaries (Source: 78 FR 20717, April 5, 2013).

Note: The Northwest Atlantic/Gulf of Mexico and Central Pacific DPSs are not listed under the ESA.

Species Description and Distribution

All hammerhead sharks belong to the family *Sphyrnidae* and are classified as ground sharks (order *Carcharhiniformes*). The hammerhead sharks are recognized by their laterally expanded head that resembles a hammer, hence the common name “hammerhead.” The scalloped hammerhead shark is distinguished from other hammerheads by a noticeable indentation on the center and front portion of the head, along with 2 more indentations on each side of this central indentation, giving the head a “scalloped” appearance. It has a broadly arched mouth, and the back of the head is slightly swept backward.

The scalloped hammerhead shark occurs over continental shelves and the shelves surrounding islands, as well as adjacent deep waters, but it is seldom found in waters cooler than 22°C (Compagno 1984; Schulze-Haugen et al. 2003). It ranges from the intertidal and surface waters to depths of up to approximately 1,475-1,675 ft (450-512 m) (Klimley 1993; Sanches 1991), with occasional dives even deeper (Jorgensen et al. 2009). It has also been documented entering enclosed bays and estuaries (Compagno 1984). All waters of the Caribbean Sea are within this DPS boundary, including the Bahamas' EEZ off the coast of Florida, the U.S. EEZ off Puerto Rico and the U.S. Virgin Islands, and Cuba's EEZ.

Scalloped hammerhead sharks are highly mobile and partly migratory, and are likely the most abundant of the hammerhead species (Maguire et al. 2006). These sharks have been observed making migrations along the edges of continents as well as between oceanic islands in tropical waters (Bessudo et al. 2011; Diemer et al. 2011; Duncan and Holland 2006; Kohler and Turner 2001). Although scalloped hammerhead sharks are highly mobile, this species rarely crosses entire oceans (Diemer et al. 2011; Duncan and Holland 2006; Kohler and Turner 2001). The median distance between mark and recapture of

3,278 tagged adult sharks along the eastern U.S. was less than 65 miles (100 km) (Kohler and Turner 2001). Tagging studies reveal the tendency for scalloped hammerhead sharks to aggregate around and travel to and from core areas or “hot spots” within locations (Bessudo et al. 2011; Duncan and Holland 2006; Hearn et al. 2010; Holland et al. 1993). However, other studies indicate they are also capable of traveling long distances (e.g., 1,206 miles [1,941 km] (Bessudo et al. 2011); 1,038 miles [1,671 km] (Kohler and Turner 2001); 390 miles [629 km] (Diemer et al. 2011).).

Both juveniles and adult scalloped hammerhead sharks occur as solitary individuals, pairs, or in schools (Compagno 1984). Adult aggregations are most common offshore over seamounts and near islands (Bessudo et al. 2011; CITES 2010; Compagno 1984; Hearn et al. 2010). Neonate and juvenile aggregations are more common in nearshore nursery habitats (Bejarano-Álvarez et al. 2011; Diemer et al. 2011; Duncan and Holland 2006). It has been suggested that juveniles inhabit these nursery areas for up to or more than 1 year as they provide valuable refuges from predation (Duncan and Holland 2006).

The scalloped hammerhead shark is a high trophic level predator (Cortés 1999) and an opportunistic feeder with a diet that includes a wide variety of bony fish, octopi/cuttlefish/squid, crabs/lobsters, and rays (Bush 2003; Compagno 1984; Júnior et al. 2009; Noriega et al. 2011).

Life History Information

The scalloped hammerhead shark gives birth to live young (i.e., “viviparous”), with a gestation period of 9-12 months (Branstetter 1987; Stevens and Lyle 1989), which may be followed by a 1-year resting period (Liu and Chen 1999). Generally, females attain maturity around 6.5-8 ft (2.0-2.5 m) TL, while males reach maturity at smaller sizes (range 4-6.5 ft [1.3-2.0 m] TL). The available information specific to the Central and Southwest Atlantic DPS indicates females attain maturity when they reach around 7.5 ft (greater than 240 cm) TL, while males reach maturity at 6-6.5 ft (1.8-2.0 m) TL (Hazin et al. 2001).

The age at maturity differs by region. In Brazil (part of the Central and Southwest Atlantic DPS), males reach sexual maturity between 6.3 and 8.1 years, females at 15.2 years (Hazin et al. 2001). However, when pupping occurs does not appear to vary by region and may be partially seasonal (Harry et al. 2011a; Harry et al. 2011b), with neonates present year round, but with abundance peaking during the spring and summer months (Adams and Paperno 2007; Bejarano-Álvarez et al. 2011; Duncan and Holland 2006; Harry et al. 2011a; Harry et al. 2011b; Noriega et al. 2011). Females move inshore to birth, with litter sizes anywhere between 1 and 41 live pups. No relationship between litter size and female shark length was identified by Hazin et al. (2001) for animals off the northeastern coast of Brazil. The DPS-specific information indicates pups are generally greater than 1.2 ft (0.38 m) at birth (Hazin et al. 2001).

While it appears that maturity, age, and growth estimates vary by region, it is unclear whether these differences are truly biological or the result of differences in the interpretations of aging methodology (Piercy et al. 2007). Scalloped hammerhead sharks develop opaque bands on their vertebrae which are used to estimate age. Assuming annual

band formation for animals in the Atlantic, and adjusting age maturity estimates from the Pacific accordingly, the average age at maturity for female scalloped hammerheads is around 12.8 years and 8.1 years for males. Based on analysis of the available data, the scalloped hammerhead shark can be characterized as a long-lived (i.e., at least 20-30 years) (Dudley and Simpfendorfer 2006), late-maturing, and relatively slow-growing species (Branstetter 1990). Within the DPS, Kotas et al. (2011) estimate the maximum age of females as 31.5 years and 29.5 years for males.

Status and Population Dynamics

Data from multiple sources indicate that the Atlantic population (including the Central and Southwest Atlantic DPS) of scalloped hammerheads has experienced severe declines over the past few decades. In a stock assessment for the scalloped hammerhead shark, (Hayes et al. 2009) concluded that the northwestern Atlantic and Gulf of Mexico scalloped hammerhead shark stock has been depleted by approximately 83% since 1981. It is likely that scalloped hammerheads in the Central and Southwest Atlantic DPS have experienced at least that level of decline since the early 1980s. Miller et al. (2014) concluded that abundance numbers for the Central and Southwest Atlantic DPS are unavailable but likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS (Models estimated the virgin population size to be between 142,000 and 169,000 individuals (range 116,000-260,000) (Hayes et al. 2009). Those models also estimated populations of 24,850-27,900 individuals in 2005 (most recent year estimated) (Hayes et al. 2009)).

It is likely that scalloped hammerheads in the Northwest Atlantic and Gulf of Mexico were overfished beginning in the early 1980s and experienced periodic overfishing from 1983-2005 (Jiao et al. 2011). Other studies have also observed similar decreases in scalloped hammerhead shark populations along the Atlantic coast. For example, Baum et al. (2003) calculated that the northwest Atlantic population of scalloped hammerhead shark has declined by 89% since 1986; however, this study is controversial due to its sole reliance on pelagic longline logbook data. Off the southeastern U.S. coast, Beerkircher et al. (2002) found significant declines in nominal CPUE for scalloped hammerhead shark between 1981-1983 (CPUE = 13.37 in (Berkeley and Campos 1988) and 1992-2000 (CPUE = 0.48).

Threats

Scalloped hammerhead sharks are both targeted and taken as bycatch in many global fisheries. They are targeted by semi-industrial, artisanal, and recreational fisheries, and caught as bycatch in pelagic longline tuna and swordfish fisheries and purse seine fisheries. There is a lack of information on the fisheries prior to the early 1970s, with only occasional mentions in historical records. Significant catches of scalloped hammerheads have gone, and continue to be, unrecorded in many countries outside the U.S. Brazil, the country that reports one of the highest scalloped hammerhead landings in South America, maintains heavy industrial fishing of this species off its coastal waters. In the late 1990s, Amorim et al. (1998) remarked that heavy fishing by longliners led to a decrease in this population off the coast of Brazil. According to the FAO global capture production database, Brazil reported a significant increase in catch of scalloped hammerhead during this period, from 30 mt in 1999 to 508 mt by 2002, before decreasing to a low of 87 mt in 2009. Information

from pelagic longline and bottom gillnet fisheries targeting several species of hammerhead sharks off southern Brazil indicates declines of more than 80% in CPUE from 2000 to 2008, with the targeted hammerhead fishery abandoned after 2008 due to the rarity of the species (FAO 2010). Scalloped hammerhead is also commonly landed by artisanal fishers in the Central and Southwest Atlantic, with concentrated fishing effort in nearshore and inshore waters, areas likely to be used as nursery grounds. In the Caribbean, specific catch and landings data are unavailable; however, scalloped hammerhead shark is often a target of artisanal fisheries off Trinidad and Tobago and Guyana, and anecdotal reports of declines in abundance, size, and distribution shifts of sharks suggest significant fishing pressure on overall shark populations in this region (Kyne et al. 2012).

The exploitation of this DPS continues to go largely unregulated. In Brazilian waters, there are very few fishery regulations that help protect hammerhead populations. For example, the minimum legal size for a scalloped hammerhead caught in Brazilian waters is approximately 24 in (60 cm) TL; however, scalloped hammerhead shark pups may range from 15-23 in (38 - 55 cm). As the pup sizes are very close to this minimum limit, the legislation is essentially ineffective, and as such, large catches of both juveniles and neonates have been documented from this region (CITES 2010; Kotas et al. 2008). Lack of enforcement of existing regulations also hamper regulatory effectiveness.

In addition, scalloped hammerheads are likely underreported in catch records as many records do not account for discards (e.g., where the fins are kept, but the carcass is discarded) or reflect dressed weights instead of live weights. Also, many catch records do not differentiate between the hammerhead species, or shark species in general, and thus species-specific population trends for scalloped hammerheads are not readily available.

Although scalloped hammerhead meat is considered essentially unpalatable (due to its high urea concentration), some countries still consume the meat domestically or trade it internationally, including Colombia, Mexico, and Uruguay (CITES 2010; Vannuccini 1999). However, it is thought that the current volume of scalloped hammerhead shark traded meat and products is insignificant when compared to the volume of its fins in international trade (CITES 2010) (Miller et al 2013).

3.2.10 Oceanic Whitetip Shark

On January 30, 2018, NMFS published a final rule that determined the oceanic whitetip shark (*Carcharhinus longimanus*) warrants listing as a threatened species (83 FR 4153). The status review report of the oceanic whitetip shark (Young et al. 2016) compiles the best available information on the status of the species as required by the ESA and assesses the current and future extinction risk for the species.

Species Description and Distribution

The oceanic whitetip shark is a large open ocean apex predatory shark found in subtropical waters around the globe. This species belongs to the family Carcharhinidae and is classified as a requiem shark (containing migratory, live-bearing sharks of the warm seas) (Order Carcharhiniformes). The oceanic whitetip belongs to the genus *Carcharhinus*, which

includes other pelagic species of sharks, such as the silky shark (*C. falciformis*) and dusky shark (*C. obscurus*), and is the only truly oceanic shark of its genus (Bonfil 2009).

The oceanic whitetip shark has a stocky build with a large rounded first dorsal fin and very long and wide paddle-like pectoral fins. The first dorsal fin is very wide with a rounded tip, originating just in front of the rear tips of the pectoral fins. The second dorsal fin originates over or slightly in front of the base of the anal fin. The species also exhibits a distinct color pattern of mottled white tips on its front dorsal, caudal, and pectoral fins with black tips on its anal fin and on the ventral surfaces of its pelvic fins. The head has a short and bluntly rounded nose and small circular eyes with nictitating membranes. The upper jaw contains broad, triangular serrated teeth, while the teeth in the lower jaw are more pointed and are only serrated near the tip. The body is grayish bronze to brown in color, but varies depending upon geographic location. The underside is whitish with a yellow tinge on some individuals. They usually cruise slowly at or near the surface with their huge pectoral fins conspicuously outspread, but can suddenly dash for a short distance when disturbed (Compagno 1984).

The oceanic whitetip shark is distributed worldwide in epipelagic tropical and subtropical waters between 30° North latitude and 35° South latitude (Baum et al. 2006). In the Western Atlantic, oceanic whitetips occur from Maine to Argentina, including the Caribbean and Gulf of Mexico.

The oceanic whitetip shark is a highly migratory species that is usually found offshore in the open ocean, on the outer continental shelf, or around oceanic islands primarily in water depths over 184 m, occurring from the surface to at least 152 m depth. EFH in the Atlantic Ocean includes localized areas in depths greater than 200 m from offshore of the North Carolina/Virginia border to the Blake Plateau. EFH in the Gulf of Mexico includes offshore habitats of the northern Gulf of Mexico at the Alabama/Florida border (i.e., the Mississippi plume seems particularly important for juveniles and adults) to offshore habitats of the western Gulf of Mexico south of eastern Texas. The entire U.S. Caribbean is considered to be EFH. Although the oceanic whitetip can be found in decreasing numbers out to latitudes of 30° N and 35° S, with abundance decreasing with greater proximity to continental shelves, it has a clear preference for open ocean waters between 10° S and 10° N (Backus et al. 1956; Bonfil et al. 2008; Compagno 1984; Strasburg 1958). The species can be found in waters between 15 °C and 28 °C, but it exhibits a strong preference for the surface mixed layer in water with temperatures above 20 °C, and is considered a surface-dwelling shark. It is however, capable of tolerating colder waters down to 7.75 °C for short periods as exhibited by brief, deep dives into the mesopelagic zone below the thermocline (>200 m), presumably for foraging (Howey-Jordan et al. 2013; Howey et al. 2016). However, exposures to these cold temperatures are not sustained (Musyl et al. 2011; Tolotti et al. 2015) and there is some evidence to suggest the species tends to withdraw from waters below 15 °C (e.g., the Gulf of Mexico in winter; Compagno 1984).

Little is known about the movement or possible migration paths of the oceanic whitetip shark. Although the species is considered highly migratory and capable of making long distance movements, tagging data provides evidence that this species also exhibits a high

degree of philopatry (i.e., site fidelity) in some locations. To date, there have been three tagging studies conducted on oceanic whitetip sharks in the Atlantic. In the Atlantic, young oceanic whitetip sharks have been found well offshore along the southeastern coast of the U.S., suggesting that there may be a nursery in oceanic waters over this continental shelf (Compagno 1984; Bonfil et al. 2008). In the southwestern Atlantic, the prevalence of immature sharks, both female and male, in fisheries catch data suggests that this area may serve as potential nursery habitat for the oceanic whitetip shark (Coelho et al. 2009; Frédou et al. 2015; Tambourgi et al. 2013; Tolotti et al. 2015). Juveniles seem to be concentrated in equatorial latitudes, while specimens in other maturational stages are more widespread (Tambourgi et al. 2013). Pregnant females are often found close to shore, particularly around the Caribbean Islands. For more information on oceanic whitetip distribution, see Young et al. (2016).

Life History Information

The oceanic whitetip shark gives birth to live young (i.e., “viviparous”). Their reproductive cycle is thought to be biennial, giving birth on alternate years, after a lengthy 10–12 month gestation period. The number of pups in a litter ranges from 1 to 14 (mean = 6), and a positive correlation between female size and number of pups per litter has been observed, with larger sharks producing more offspring (Bonfil et al. 2008; Compagno 1984; IOTC 2014; Seki et al. 1998). Age and length of maturity estimates are slightly different depending on geographic location. In the Southwest Atlantic, age and length of maturity in oceanic whitetips was estimated to be 6–7 years and 180–190 cm TL, respectively, for both sexes (Lessa et al. 1999).

Historically, the maximum length effectively measured for the oceanic whitetip was 350 cm TL (Bigelow and Schroder 1948 cited in Lessa et al. 1999), with “gigantic individuals” perhaps reaching 395 cm TL (Compagno 1984), though Compagno’s length seems to have never been measured (Lessa et al. 1999). In contemporary times, Lessa et al. (1999) recorded a maximum size of 250 cm TL in the Southwest Atlantic, and estimated a theoretical maximum size of 325 cm TL (Lessa et al. 1999), but the most common sizes are below 300 cm TL (Compagno 1984). The oceanic whitetip has an estimated maximum age of 17 years, with confirmed maximum ages of 12 and 13 years in the North Pacific and South Atlantic, respectively (Seki et al. 1998; Lessa et al. 1999). However, other information from the South Atlantic suggests the species likely lives up to ~20 years old based on observed vertebral ring counts (Rodrigues et al. 2015). Growth rates (growth coefficient, K) have been estimated similarly for both sexes and range from 0.075–0.099 in the Southwest Atlantic to 0.0852–0.103 in the North Pacific (Joung et al. 2016; Lessa et al. 1999; Seki et al. 1998). Using life history parameters from the Southwest Atlantic, (Cortés et al. 2010; Cortés et al. 2012) estimated productivity of the oceanic whitetip shark, determined as intrinsic rate of population increase (r), to be 0.094–0.121 per year (median). Overall, the best available data indicate that the oceanic whitetip shark is a longlived species (at least 20 years) and can be characterized as having relatively low productivity.

To date, only two studies have been conducted on the genetics and population structure of the oceanic whitetip shark, which suggest there may be some genetic differentiation between various populations of the species. Overall, the data showing population structure

within the Atlantic relies solely on mitochondrial DNA and does not reflect male mediated gene flow. Thus, while the current data supports three maternal populations within the Atlantic, information regarding male mediated gene flow would provide an improved understanding of the fine-scale genetic structuring of oceanic whitetip in the Atlantic. On the other hand, both mitochondrial DNA and nuclear microsatellite data analyses support at least two global genetic stocks. However, the data from these studies are preliminary, and it is likely that additional population structure within and between oceans will be discovered with additional samples and analyses.

Oceanic whitetip sharks are high trophic-level predators in open ocean ecosystems feeding mainly on teleosts and cephalopods ((Backus et al. 1956; Bonfil et al. 2008), but studies have also reported that they consume sea birds, marine mammals, other sharks and rays, molluscs, crustaceans, and even garbage (Compagno 1984; Cortés 1999). Backus et al. (1956) recorded various fish species in the stomachs of oceanic whitetip sharks, including blackfin tuna, barracuda, and white marlin. Based on the species' diet, the oceanic whitetip has a high trophic level, with a score of 4.2 out of a maximum 5.0 (Cortés 1999). The available evidence also suggests that oceanic whitetip sharks are opportunistic feeders.

Status and Population Dynamics

Oceanic whitetip sharks can be found worldwide, with no present indication of a range contraction. While a global population size estimate or trend for the oceanic whitetip shark is currently unavailable, numerous sources of information, including the results of a recent stock assessment and several other abundance indices (e.g., trends in occurrence and composition in fisheries catch data, CPUE, and biological indicators) were available to infer and assess current regional abundance trends of the species. Given the available data, and the fact that the available assessments were not conducted prior to the advent of industrial fishing (and thus not from virgin biomass), the exact magnitude of the declines and current abundance of the global population are unknown. The oceanic whitetip shark was historically one of the most abundant and ubiquitous shark species in tropical seas around the world; however, numerous lines of evidence suggest declines greater than 70-80% in most areas throughout its range, and this species likely continues to experience abundance declines of varying magnitude globally.

In the Northwest Atlantic, the oceanic whitetip shark was described historically as widespread, abundant, and the most common pelagic shark in the warm parts of the North Atlantic (Backus et al. 1956). Recent information, however, suggests the species is now relatively rare in this region.

Several studies have been conducted in this region to determine trends in abundance of various shark species, including the oceanic whitetip shark, and these studies have shown significant declines in abundance. The proposed listing rule provides more detail on the varying estimates on the severity of the declines (81 FR 96304, December 29, 2016). Relative abundance of oceanic whitetip shark may have stabilized in the Northwest Atlantic since 2000 and in the Gulf of Mexico/Caribbean since the late 1990s at a significantly diminished abundance (Young et al. 2016).

Threats

Currently, the most significant threat to oceanic whitetip sharks is mortality in commercial fisheries, largely driven by demand of the international shark fin trade, bycatch related mortality, as well as illegal, unreported, and unregulated (IUU) fishing. Although generally not targeted, oceanic whitetip sharks are frequently caught as bycatch in many fisheries, including pelagic longline fisheries targeting tuna and swordfish, purse seine, gillnet, and artisanal fisheries. Oceanic whitetip sharks are also a preferred species for their large, morphologically distinct fins, as they obtain a high price in the Asian fin market. The oceanic whitetip shark's vertical and horizontal distribution significantly increases its exposure to industrial fisheries, including pelagic longline and purse seine fisheries operating within the species' core tropical habitat throughout its global range.

In addition to declines in oceanic whitetip catches throughout its range, there is also evidence of declining average size over time in some areas, and this is a concern for the species' status given evidence that litter size is positively correlated with maternal length. Such extensive declines in the species' global abundance, the ongoing threat of overutilization, and the species' slow growth and relatively low productivity, makes them generally vulnerable to depletion and potentially slow to recover from overexploitation. Related to this, the low genetic diversity of oceanic whitetip is also cause for concern and a viable risk over the foreseeable future for this species. Loss of genetic diversity can lead to reduced fitness and a limited ability to adapt to a rapidly changing environment. The biology of the oceanic whitetip shark indicates that it is likely to be a species with low resilience to fishing and minimal capacity for compensation (Rice and Harley 2012).

Available information does not indicate that destruction, modification or curtailment of the species' habitat or range, disease or predation, or other natural or manmade factors are operative threats on this species (81 FR 96304, December 29, 2016).

3.2.11 Giant Manta Ray

On January 22, 2018, NOAA Fisheries published a final rule listing the giant manta ray (*Manta birostris*) as threatened under the ESA effective February 21, 2018 (83 FR 2916). The status review report of the giant manta ray (Miller and Klimovich 2017) compiles the best available information on the status of the species as required by the ESA and assesses the current and future extinction risk for the species.

Species Description and Distribution

The genus *Manta* includes the giant manta ray (*Manta birostris*) and the reef manta ray (*Manta alfredi*). Historically, the genus *Manta* was considered monotypic and was categorized as a single species, *M. birostris*. The genus was re-evaluated and split into two species: *M. birostris* and *M. alfredi* (Marshall et al. 2009). The two species are distinguished based on physical characters such as coloration, dentition, denticles, spine morphology, size at maturity, and maximum disc width (DW) (Marshall et al. 2009). Genetic evidence further confirmed the existence of the two separate species (Kashiwagi et al. 2008; Ito and Kashiwagi 2010). In the Atlantic and Gulf of Mexico, a third, undescribed species may be distinct from *M. birostris*, but further examination of specimens is necessary to clarify the taxonomic status of this variant manta ray (Marshall

et al. 2009). At present there is not enough empirical evidence to warrant the separation of a third species of *Manta*.

The giant manta ray is the largest living ray, with a wingspan reaching a width of up to 9 m (29 ft). The distance over this wingspan is termed disc width (DW). The giant manta rays have two distinct color types: chevron (mostly black back dorsal side and white ventral side) and black (almost completely black on both ventral and dorsal sides). Most of the chevron variants have a black dorsal surface and a white ventral surface with distinct patterns on the underside that can be used to identify individuals (Miller and Klimovich 2017). While these markings are assumed to be permanent, there is some evidence that the pigmentation pattern of *M. birostris* may actually change over the course of development (based on observation of two individuals in captivity), and thus caution may be warranted when using color markings for identification purposes in the wild (Ari 2015). *Manta* species are distinguished from other *Mobula* in that they tend to be larger, with a terminal mouth, and have long cephalic lobes (Evgeny 2010), which are extensions of the pectoral fins that funnel water into the mouth.

The giant manta ray is a large pelagic filter feeder found in tropical and subtropical seas. These slow-growing, migratory animals are circumglobal with fragmented populations. They are found worldwide in tropical, subtropical, and temperate bodies of water. On the east coast of the U.S., their range occurs as far north as New Jersey and as far south as Florida. Their range extends from Florida south past the Caribbean islands and along the east coast of South America. Preliminary research suggests that the shallow, nearshore waters off the Atlantic coast of southeastern Florida may be a nursery ground for juvenile giant manta ray (J. Pate, unpublished data). Personal observation during aerial surveys conducted off of St. Augustine, Florida, from 2009-2012, F. Young (pers. comm. 2017, as cited in Miller and Klimovich 2017) noted vast schools of giant manta rays, with over 500 manta rays observed per 6-8 hour day of aerial survey. In addition, recent research at the Flower Garden Banks National Marine Sanctuary indicates that the sanctuary and the surrounding banks in the northwest Gulf of Mexico is a nursery habitat for juvenile *M. birostris* (Stewart et al. in review). However, researchers are actively trying to determine whether the giant manta rays in this area are only *M. birostris* individuals or potentially also comprise individuals of an undescribed species (Marshall et al. 2009; Hinojosa-Alvarez et al. 2016, Stewart et al. in review).

The giant manta ray is considered to be a migratory species, with satellite tracking studies using pop-up satellite archival tags registering movements of the giant manta ray from Mozambique to South Africa (a distance of 1,100 km), from Ecuador to Peru (190 km), and from the Yucatan, Mexico into the Gulf of Mexico (448 km) (Marshall et al. 2011). Although recent tagging data suggest that while the species may be capable of occasional long-distance movements, it may be more typical for these species to exhibit a high degree of residency (Miller and Klimovich 2017). Site fidelity has been shown to specific regions, and habitats within them, such as cleaning stations and feeding sites. Preliminary satellite tracking studies and international photo-identification matching projects have suggested a low degree of interchange between populations (Marshall et al. 2011).

Giant manta rays are seasonal visitors along productive coastlines with regular upwelling, in oceanic island groups, and near offshore pinnacles and seamounts. The timing of these visits varies by region and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior. They have also been observed in estuarine waters near oceanic inlets, with use of these waters as potential nursery grounds (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al. 2015). Giant manta rays can be found in cool water, as low as 19°C, although temperature preference appears to vary by region; along the U.S. east coast they are commonly found in waters from 19 to 22°C. (Freedman and Roy 2012; Miller and Klimovich 2017).

Although giant manta rays are considered oceanic and solitary, they have been observed congregating at cleaning sites at offshore reefs and feeding in shallow waters during the day at depths less than 10 m (O'Shea et al. 2010; Marshall et al. 2011; Rohner et al. 2013). This species appears to exhibit a high degree of plasticity in terms of their use of depths within their habitat. Tagging studies have shown that the giant manta rays do conduct night descents from 200-450m depths (Rubin et al. 2008; Stewart et al. 2016) but are capable of diving to depths exceeding 1,000 m (A. Marshall et al. unpubl. data 2011, cited in Marshall et al. (2011)). The species has a rete mirabile cranica¹⁷ as a counter-current heat exchanger around the brain that possibly facilitates its use of these cooler habitats (Alexander 1996). Stewart et al. (2016) found diving behavior may be influenced by season, and more specifically, shifts in prey location associated with the thermocline, with tagged giant manta rays (n=4) observed spending a greater proportion of time at the surface from April to June and in deeper waters from August to September.

Life History Information

The giant manta ray gives birth to live young (i.e., “viviparous”). Manta rays are slow to mature and also have very low fecundity and typically give birth to only one pup every two to three years. Gestation lasts from approximately 10-14 months. Females are only able to produce between 5 and 15 pups in a lifetime (CITES 2013; Miller and Klimovich 2017). This species has one of the lowest maximum population growth rates of all elasmobranchs (Dulvy et al. 2014; Miller and Klimovich 2017). The giant manta rays generation time (based on *M. alfredi* life history parameters) is estimated to be 25 years (Miller and Klimovich 2017).

Although giant manta rays have been reported to live at least 40 years, not much is known about their growth and development. Maturity is thought to occur between 8-10 years of age on average and the status review further discusses the range of maturity estimates (Miller and Klimovich 2017). Size at maturity varies slightly throughout their range, with males estimated to mature when smaller than females around 3.8-4 m DW, and females at around 4.1-4.7 m DW (Miller and Klimovich 2017).

¹⁷ Arete mirabile cranica is a massive arterial network grossly divisible into a “caudal RMC” supplying blood to the brain, and an expanded, more complex “precerebral RMC” nested within the large cranial cavity rostral to the telencephalon.

Giant manta rays feed primarily on planktonic organisms, but some studies have documented consumption of small fishes (Miller and Klimovich 2017). For example, Rohner et al. (2017a) documented two species of myctophid fishes in the stomach contents of *M. birostris* in the Bohol Sea (Philippines). However, planktonic organisms appear to comprise the majority of the diet. Clark (2010) suggests that the larger *M. birostris* may forage in less productive pelagic waters and conduct seasonal migrations following prey abundance. While it was previously assumed, based on field observations, that giant manta rays feed predominantly during the day on surface zooplankton, results from recent studies (Couturier et al. 2013; Burgess et al. 2016) indicate that these feeding events are not an important source of the dietary intake. Burgess et al. (2016) used stable isotope analysis of muscle tissues of individuals collected off Ecuador and surface zooplankton to examine the giant manta ray diet. The authors found that, on average, mesopelagic sources contributed 73% to the giant manta ray's diet, compared to 27% for surface zooplankton (Burgess et al. 2016). Overall, studies indicate that giant manta rays have a more complex depth profile of their foraging habitat than previously thought, and may actually be supplementing their diet with the observed opportunistic feeding in near-surface waters (Couturier et al. 2013; Burgess et al. 2016). When feeding, mantas hold their cephalic fins in an "O" shape and open their mouth wide which creates a funnel that pushes water and prey through their mouth and over their gill rakers. They use many different types of feeding strategies, such as barrel rolling (doing somersaults over and over again) and creating feeding chains with other mantas to maximize prey intake.

Status and Population Dynamics

There are no current or historical estimates of their global abundance, with most estimates of subpopulations based on anecdotal diver or fisherman observations. CITES (2013) reports that only ten populations of giant manta rays have been actively studied, 25 other aggregations have been anecdotally identified, and all other sightings are rare, the total global population may be small. Subpopulation abundance estimates range between 100 and 1,500 individuals, but are anecdotal and subject to bias (Miller and Klimovich 2017). The largest subpopulations and records of individuals come from the Indo-Pacific and eastern Pacific. Miller and Klimovich (2017) concluded that giant manta rays are at risk throughout a significant portion of their range, due in large part to the observed declines in the Indo-Pacific. There have been decreases in landings of up to 95 % in the Indo-Pacific; such declines have not been observed in other subpopulations such as Mozambique and Ecuador. Atlantic populations are likely small and sparsely distributed. There have been reports of more than 90 individuals off the east coast of Florida. There are records of over 70 individuals from the Flower Garden Banks Marine Sanctuary in the Gulf of Mexico. Overall, given their life history traits, particularly their low reproductive output, giant manta ray populations are inherently vulnerable to depletions, with low likelihood of recovery.

Threats

The most significant threat to the species is overutilization for commercial purposes. The species is targeted by some fisheries, and also caught as bycatch. The internal gill raker trade has been increasing and the demand for the gills of manta and other rays has risen dramatically in Asian markets. They are most susceptible to purse-seine and artisanal

gillnet fisheries in the Indo-Pacific and eastern Pacific and inadequate regulatory mechanisms to protect them from the heavy fishing pressure and related mortality in these waters outside of U.S. jurisdiction. In areas where the species is actively targeted or caught as bycatch (e.g., Philippines, Mexico, Sri Lanka, Indonesia), populations appear to be decreasing (Miller and Klimovich 2017). Take and trade in U.S. waters were not identified as significant threats. In areas where the species is not subject to fishing, population abundance may be stable (Miller and Klimovich 2017). In U.S. waters the giant manta ray is bycaught in several fisheries; however the status review and the final rule found that U.S. fishery bycatch levels were determined to only have a minimal impact on the status of the giant manta ray.

While the species is not targeted by fisheries within the U.S. EEZ, mantas are subjected to pressure ship strikes, entanglement, and marine debris “ghost” fishing gear (Deakos et al. 2011) that potentially contribute to increased mortality rates. Because giant manta ray are observed close to shore, at nearshore aggregation sites, and ocean inlets that are sometimes in areas of high maritime traffic, manta rays are at potential risk of being struck and killed by vessels. In addition, derelict fishing lines and other commercial gear is an entanglement risk and can cause significant external damage, negatively affecting their ability to swim and feed. Mooring and boat anchor line entanglement may also wound manta rays or cause them to drown (Deakos et al. 2011; Heinrichs et al. 2011). For example, in a Maui, Hawaii, *M. alfredi* population (n= 290 individuals), Deakos et al. (2011) observed that 1 out of 10 reef manta rays had an amputated or disfigured non-functioning cephalic fin, likely a result of line entanglement. Internet searches also reveal photographs of mantas with injuries consistent with vessel strikes and line entanglements, and manta researchers report that such injuries may affect manta fitness in a significant way (The Hawaii Association for Marine Education and Research Inc. 2005; Deakos et al. 2011; Heinrichs et al. 2011; Couturier et al. 2012; CMS 2014; Germanov and Marshall 2014; Braun et al. 2015). Similarly, researchers along the east coast of Florida report sightings of giant manta rays either foul-hooked or entangled in fishing line as being relatively common (J. Pate, unpublished data). Personal observations and photographs of giant manta rays with injuries that are consistent with vessel strike have also been reported along Florida’s east coast (C. Horn pers. comm. 2018). However, there is very little quantitative information on the frequency of these occurrences and no information on the impact of these injuries on the overall health of the population.

Since giant manta rays are filter feeders, ingestion of plastics is common and has been thoroughly documented (Miller and Klimovich 2017).

4.0 Environmental Baseline

This section describes the effects of past and ongoing human and natural factors contributing to the current status of the species that are likely to be adversely affected by the action (i.e., sea turtles, smalltooth sawfish, Atlantic sturgeon, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray), their habitats (including designated critical habitat), and ecosystem within the action area, without the additional effects of the proposed action. In the case of ongoing actions, this section includes the effects that may contribute to the projected future status of the species, their habitats and ecosystems. The environmental baseline describes the species' and habitat health, based on information available at the time of this consultation.

By regulation (50 CFR 402.02), the environmental baseline refers to the condition of the listed species in the action area, without the consequences to the listed species caused by the proposed action. The environmental baselines for Biological Opinions include the past and present impacts of all state, federal, or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area of the consultation at issue that have already undergone formal or early Section 7 consultation (as defined in 50 CFR 402.11), as well as the impact of state or private actions that are contemporaneous with the consultation in process.

Focusing on the impacts of the activities in the action area specifically allows us to assess the prior experience and state (or condition) of the endangered and threatened individuals, and areas of designated critical habitat that occur in an action area, and that will be exposed to effects from the action under consultation. This is important because, in some states or life history stages, or areas of their ranges, listed individuals or critical habitat features will commonly exhibit, or be more susceptible to, adverse responses to stressors than they would be in other states, stages, or areas within their distributions. These localized stress responses or stressed baseline conditions may increase the severity of the adverse effects expected from the proposed action.

4.1 Status of Species in the Action Area

As stated in Section 2.11 (Action Area), the proposed action occurs in the U.S. EEZ in the Atlantic Ocean, the Gulf of Mexico, and the Caribbean Sea.

Sea Turtles

The 6 species of sea turtles that occur in the action area—the North Atlantic (NA) and South Atlantic (SA) DPSs of green sea turtle, hawksbill sea turtle, leatherback sea turtle, the Northwest Atlantic (NWA) DPS of loggerhead sea turtle, and Kemp's ridley sea turtle—are all highly migratory. Given the large size of the action area, all sea turtle life stages, and associated behaviors (except nesting) occur in the action area. Therefore, the status of each of the sea turtles in the action area, as well as the threats to these species, are best reflected in their range-wide statuses and supported by the species accounts in Section 3 (Status of Species).

Smalltooth Sawfish

Smalltooth sawfish greater than 200 cm TL may be found the action area, off Florida, throughout the year. The status of smalltooth sawfish in the action area, as well as the threats to this species, is reflected and supported by the species account in Section 3 (Status of the Species).

Atlantic Sturgeon

All of the Atlantic sturgeon DPSs (i.e., the South Atlantic, Carolina, Chesapeake Bay, New York Bight, and Gulf of Maine DPSs) may be found in the action area, off the East coast of Florida, from Cape Canaveral, Florida and north to the northern limit of the action area. The status of the DPSs in the action area, as well as the threats to these DPSs, is reflected in and supported by the species accounts in Section 3 (Status of the Species).

Scalloped Hammerhead Shark— Central and Southwest Atlantic DPS

The Central and Southwest Atlantic DPS of scalloped hammerhead sharks may be found in the southern portion of the action area (Caribbean). The status of species in the action area, as well as the threats to this species, is reflected in and supported by the species account in Section 3 (Status of the Species).

Oceanic Whitetip Shark

Given the large size of the action area and the wide range of the oceanic whitetip shark, oceanic whitetip sharks could occur throughout the action area. Therefore, the status of oceanic whitetips sharks in the action area, as well as the threats to this species, is reflected in and supported by the species accounts in Section 3 (Status of Species).

Giant Manta Ray

Within the action area, the giant manta ray may be found in the Gulf of Mexico, the Caribbean Sea, and off the Atlantic East coast to as far north as New Jersey. The status of giant manta rays in the action area, as well as the threats to the species, is reflected in and supported by the species accounts in Section 3.

4.2 Factors Affecting Sea Turtles in the Action Area

The following analysis examines actions that may affect sea turtle species, namely the NWA DPS of loggerhead sea turtle, leatherback sea turtle, Kemp's ridley sea turtle, the NA and SA DPSs of green sea turtle, and hawksbill sea turtle, and their environments specifically within the action area. Sea turtles found in the immediate project area may travel widely throughout the Atlantic, Gulf of Mexico, and Caribbean Sea, and individuals found in the action area can potentially be affected by activities anywhere within this wide range. Impacts outside of the action area are discussed and incorporated as part of the overall status of the species as detailed in Status of Species section, above. The activities that shape the environmental baseline for sea turtles in the action area of this consultation are primarily fisheries, vessel operations, permits allowing take under the ESA, military activities, dredging, marine pollution, coastal development, and climate change.

4.2.1 Federal Actions

NMFS has undertaken a number of Section 7 consultations to address the effects of federally authorized fisheries and other federal actions on threatened and endangered sea turtle species, and, when appropriate, has authorized the incidental taking of these species in association with these actions, subject to certain conditions. Each of those consultations sought to minimize the adverse impacts of the action on sea turtles. Similarly, NMFS has undertaken recovery actions under the ESA that also seek to address sea turtle captures/interactions resulting from federal activities. As stated in Section 4, the summary below of federal actions and the effects these actions have had on sea turtles includes only those federal actions in the action area that have already concluded or are currently undergoing formal Section 7 consultation or that have undergone early section 7 consultation.

4.2.1.1 Federal Fisheries

Threatened and endangered sea turtles are adversely affected by several types of fishing gears used throughout the action area. Gillnet, longline, other types of hook-and-line gear, trawl gear, and pot fisheries have all been documented as interacting with sea turtles. Available information suggests sea turtles can be captured in any of these gear types when the operation of the gear overlaps with the distribution of sea turtles. For all fisheries for which there is a federally approved FMP or other federal action to manage the fishery, impacts have been evaluated under Section 7. Formal Section 7 consultations have been conducted concerning effects of the following fisheries, which occur at least in part within the action area. These fisheries have been found to be likely to adversely affect threatened and endangered sea turtles. An Incidental Take Statement (ITS) has been issued for the take of sea turtles in each of these fisheries and the take numbers depict the relative impact of each fishery on sea turtles from the data of the ITS forward in time (Appendix A). A brief summary of each fishery and its impacts on sea turtles is provided below, but more detailed information can be found in the respective Biological Opinions.

Southeastern Shrimp Trawl Fisheries

NMFS has prepared Opinions on shrimp trawling numerous times over the years (most recently 20012 and 2014). The consultation history is closely tied to the lengthy regulatory history governing the use of TEDs and a series of regulations aimed at reducing potential for incidental mortality of sea turtles in commercial shrimp trawl fisheries. By the late 1970s, there was evidence that thousands of sea turtles were being killed annually in the Southeast (Henwood and Stuntz 1987). In 1990, the National Research Council concluded the Southeast shrimp trawl fishery affected more sea turtles than all other activities combined and was the most significant anthropogenic source of sea turtle mortality in the U.S. waters, in part due to the high reproductive value of turtles taken in this fishery (NRC 1990).

The level of annual mortality described in (NRC 1990) is believed to have continued until 1992-1994, when U.S. law required all shrimp trawlers in the Atlantic and Gulf of Mexico to use TEDs, allowing at least some sea turtles to escape nets before drowning (NMFS

2002a).¹⁸ TEDs approved for use have had to demonstrate 97% effectiveness in excluding sea turtles from trawls in controlled testing. These regulations have been refined over the years to ensure that TED effectiveness is maximized through proper placement and installation, configuration (e.g., width of bar spacing), flotation, and more widespread use.

Despite the apparent success of TEDs for some species of sea turtles (e.g., Kemp's ridleys), it was later discovered that TEDs were not adequately protecting all species and size classes of sea turtles. Analyses by Epperly and Teas (2002) indicated that the minimum requirements for the escape opening dimension in TEDs in use at that time were too small for some sea turtles and that as many as 47% of the loggerheads stranding annually along the Atlantic and Gulf of Mexico were too large to fit the existing openings. On December 2, 2002, NMFS completed an Opinion on shrimp trawling in the southeastern U.S. (NMFS 2002a) under proposed revisions to the TED regulations requiring larger escape openings (68 FR 8456, February 21, 2003). This Opinion determined that the shrimp trawl fishery under the revised TED regulations would not jeopardize the continued existence of any sea turtle species. The determination was based in part on the Opinion's analysis that showed the revised TED regulations were expected to reduce shrimp trawl related mortality by 94% for loggerheads and 97% for leatherbacks. In February 2003, NMFS implemented the revisions to the TED regulations.

On May 9, 2012, NMFS completed an Opinion that analyzed the implementation of the sea turtle conservation regulations that contain TED provisions, and the operation of the Southeast U.S. shrimp fisheries in federal waters under the Magnuson-Stevens Act (NMFS 2012b). The Opinion also considered a proposed amendment to the sea turtle conservation regulations to withdraw the alternative tow time restriction at 50 CFR 223.206(d)(2)(ii)(A)(3) for skimmer trawls, pusher-head trawls, and wing nets (butterfly trawls) and instead require all of those vessels to use TEDs. The Opinion concluded that the proposed action was not likely to jeopardize the continued existence of any sea turtle species. An ITS was provided that used anticipated trawl effort and fleet TED compliance (i.e., compliance resulting in overall average sea turtle catch rates in the shrimp otter trawl fleet at or below 12%) as surrogates for sea turtle takes. On November 21, 2012, NMFS determined that a Final Rule requiring TEDs in skimmer trawls, pusher-head trawls, and wing nets was not warranted and withdrew the proposal. The decision to not implement the Final Rule created a change to the proposed action analyzed in the 2012 Opinion. Consequently, NMFS reinitiated consultation on November 26, 2012. Consultation was completed in April 2014 and determined the implementation of the sea turtle conservation regulations and the operation of the Southeast U.S. shrimp fisheries in federal waters under the Magnuson-Stevens Act was not likely jeopardize the continued existence of any sea turtle species. The ITS maintained the use of anticipated trawl effort and fleet TED compliance as surrogates for numerical sea turtle takes. Appendix A reports the takes currently authorized for the fishery.

¹⁸ TEDs were mandatory on all shrimping vessels; however, certain shrimpers (e.g., fishers using skimmer trawls or targeting bait shrimp) could operate without TEDs if they agreed to follow specific tow time restrictions.

Atlantic HMS Pelagic Longline Fishery

The Atlantic pelagic longline fishery for swordfish and tuna has been known to incidentally capture and kill large numbers of loggerhead and leatherback sea turtles. U.S. pelagic longline fishermen began targeting highly migratory species in the Atlantic Ocean in the early 1960s. The fishery is comprised of five relatively distinct segments, including: the Gulf yellowfin tuna fishery (the only segment in our action area); southern Atlantic (Florida East Coast to Cape Hatteras) swordfish fishery; Mid-Atlantic and New England swordfish and bigeye tuna fishery; U.S. Atlantic Distant Water swordfish fishery; and the Caribbean tuna and swordfish fishery. Pelagic longlines targeting yellowfin tunas in the Gulf are set in the morning (pre-dawn) in deep water and hauled in the evening. The fishery mainly interacts with leatherback sea turtles and pelagic juvenile loggerhead sea turtles, thus, younger, smaller loggerhead sea turtles than some of the other fisheries described in this environmental baseline.

Over the past two decades, NMFS has conducted numerous consultations on this fishery, some of which required RPAs to avoid jeopardy of loggerhead and/or leatherback sea turtles. The estimated historical total number of loggerhead and leatherback sea turtles caught between 1992-2002 (all geographic areas) is 10,034 loggerhead and 9,302 leatherback sea turtles of which 81 and 121 were estimated to be dead when brought to the vessel (NMFS 2004c). This does not account for post-release mortalities, which were likely substantial.

NMFS issued the most recent Opinion considering the effects of the HMS pelagic longline fishery managed under the 2006 Consolidated HMS FMP (as amended) in 2004 (NMFS 2004). This Opinion followed from a reinitiation of consultation that was undertaken because the authorized number of incidental takes for loggerheads and leatherbacks sea turtles were exceeded. The resulting 2004 Opinion stated the operation of the Atlantic pelagic longline fishery under the 1999 HMS FMP as proposed was likely to jeopardize the continued existence of leatherback sea turtles, but RPAs were identified allowing for the operation of the pelagic longline fishing that would not jeopardize leatherback sea turtles. Appendix A reports the takes currently authorized for this fishery.

As described in more detail in Section 4.2.5, below, as a result of the 2004 Opinion, on July 6, 2004, NMFS published a Final Rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic pelagic longline fishery (69 FR 40734). Swimmer et al. (2017) found bycatch rates of leatherback and loggerhead sea turtles declined by 40 and 61 percent for leatherback and loggerhead sea turtles, respectively, following these regulations. Within the NED area alone, where additional restrictions include a large circle hook (18/0) and limited use of squid bait, rates declined by 64 and 55% for leatherback and loggerhead turtles, respectively (Swimmer et al. 2017).

On March 31, 2014, the HMS Management Division requested additional consultation on the pelagic longline fishery based on information indicating that the net mortality rate and total mortality estimates for leatherback sea turtles specified in the reasonable and prudent alternative were exceeded (although the take level specified in the incidental take statement has not been exceeded), changes in information about leatherback and loggerhead sea turtle

populations, and new information about sea turtle mortality associated with PLL gear. That consultation is on-going.

Atlantic HMS Fisheries for Shark, Swordfish, Tuna, and Billfish, Excluding the Pelagic Longline Fishery

These non-pelagic longline fisheries are the subject of this consultation, and their operation to date has affected and is part of the environmental baseline for sea turtles in the action area for this consultation. This opinion evaluates the operation of those fisheries, i.e., the future effects of those fisheries on sea turtles, and other species.

As described in Section 1, NMFS consulted on the effects of all of the subject fisheries in 2001 (NMFS 2001) and has formally consulted 3 more times on the effects of HMS shark fisheries on sea turtles (NMFS 2003a; NMFS 2008b; NMFS 2012c). The most recent ESA Section 7 consultation on the operation of Atlantic shark fisheries was completed on December 12, 2012. That Opinion considered the operation of those fisheries as modified by Amendments 3 and 4 to the Consolidated HMS FMP (NMFS 2012c). In that consultation, we concluded the proposed action was not likely to jeopardize the continued existence of sea turtles. An ITS was provided authorizing takes. Appendix A reports the takes authorized for the fishery prior to completion of this consultation.

Gulf of Mexico Reef Fish Fishery

The Gulf of Mexico reef fish fishery uses two basic types of gear: spear or powerhead, and hook-and-line gear. Hook-and-line gear used in the fishery includes both commercial bottom longline and commercial and recreational vertical line (e.g., handline, bandit gear, rod-and-reel). Prior to 2008, the reef fish fishery was believed to have a relatively moderate level of sea turtle bycatch attributed to the hook-and-line component of the fishery (i.e., approximately 107 captures and 41 mortalities annually, all species combined, for the entire fishery) (NMFS 2005c). In 2008, SEFSC observer programs and subsequent analyses indicated that the overall amount and extent of incidental take for sea turtles specified in the incidental take statement of the 2005 opinion on the reef fish fishery had been severely exceeded by the bottom longline component of the fishery, with estimates more than three times the authorized levels. The west Florida shelf is an important sea turtle foraging habitat. Individual sea turtles incidentally caught by the longline component of the fishery are sexually immature juveniles and mature adult loggerhead sea turtles that have high reproductive potential.

In response, NMFS published an emergency rule prohibiting the use of bottom longline gear in the reef fish fishery shoreward of a line approximating the 50-fathom depth contour in the eastern Gulf of Mexico, essentially closing the bottom longline sector of the reef fish fishery in the eastern Gulf of Mexico for six months pending the implementation of a long-term management strategy. The Gulf of Mexico Fishery Management Council (GMFMC) developed a long-term management strategy via a new amendment (Amendment 31 to the Reef Fish FMP). The amendment included a prohibition on the use of bottom longline gear in the Gulf of Mexico reef fish fishery, shoreward of a line approximating the 35-fathom contour east of Cape San Blas, Florida, from June through August; a reduction in the number of bottom longline vessels operating in the fishery via an endorsement

program; and a restriction on the total number of hooks that may be possessed onboard each Gulf of Mexico reef fish bottom longline vessel to 1,000, only 750 of which may be rigged for fishing.

On October 13, 2009, SERO completed an opinion that analyzed the expected effects of the operation of the Gulf of Mexico reef fish fishery under the changes proposed in Amendment 31 (NMFS-SEFSC 2009c). The opinion concluded that sea turtle takes would be substantially reduced compared to the fishery as it was previously prosecuted, and that operation of the fishery would not jeopardize the continued existence of any sea turtle species. Amendment 31 was implemented on May 26, 2010. In August 2011, consultation was reinitiated to address the DWH oil release event and potential changes to the environmental baseline. Reinitiation of consultation was not related to any material change in the fishery itself, violations of any terms and conditions of the 2009 opinion, or exceedance of the incidental take statement. The resulting September 11, 2011, opinion concluded the operation of the Gulf reef fish fishery is not likely to jeopardize the continued existence of any listed sea turtles, and an ITS was provided (NMFS 2011). Appendix A reports the takes currently authorized for the fishery.

South Atlantic Snapper-Grouper Fishery

NMFS most recently prepared an Opinion on the South Atlantic Snapper-Grouper Fishery in 2016. The South Atlantic Snapper-Grouper Fishery uses spear and powerheads, black seabass (BSB) pot, and hook-and-line gear. Hook-and-line gear used in the fishery includes commercial bottom longline gear and commercial and recreational vertical line gear (e.g., handline, bandit gear, and rod-and-reel). The 2016 consultation concluded the operation of the fishery was not likely to jeopardize the continued existence of any listed species. Appendix A reports the takes authorized for this fishery.

Caribbean Reef Fish Fishery

NMFS completed an ESA Section 7 consultation on the Caribbean reef fish fishery on October 4, 2011. The reef fish fishery in waters around Puerto Rico and the USVI uses pots and traps, hook and line, longline, and spearguns. The fishery targets snapper and groupers, as well as herbivorous fish (i.e., parrotfish and surgeonfish). The Opinion concluded that the fishery was likely to adversely affect green, hawksbill, and leatherback sea turtles via vessel strikes and entanglements in fishing gear, but it would not jeopardize their continued existence. An ITS was issued authorizing incidental take. Appendix A reports the takes currently authorized for the fishery.

Caribbean Spiny Lobster Fishery

The spiny lobster fishery in waters around Puerto Rico and the USVI occurs with pots and traps, and hand-harvest. Due to the predominance of fishable habitat in state waters, it is assumed that most of the commercial harvest occurs in state waters, but fishery statistics do not allow accurate separation of harvest in the EEZ from harvest in state waters (Matos-Caraballo 2002). NMFS completed a formal consultation on the fishery on December 12, 2011 (NMFS 2011d). The Opinion concluded that the operation of the fishery was likely to adversely affect leatherback, green, and hawksbill sea turtles. Those effects were not

likely to jeopardize the continued existence of any species, and an ITS for sea turtles was issued. Appendix A reports the takes currently authorized for the fishery.

Gulf of Mexico and South Atlantic Spiny Lobster Fishery

NMFS completed a Section 7 consultation on the Gulf of Mexico and South Atlantic Spiny Lobster FMP on August 27, 2009 (NMFS 2009e). The commercial component of the fishery consists of diving, bully net and trapping sectors; recreational fishers are authorized to use bully net, and hand-harvest gears. Of the gears used, only traps are expected to result in adverse effects on sea turtles. The consultation determined the operation of the fishery would not jeopardize any sea turtle species. An ITS was issued for takes in the commercial trap sector of the fishery. Appendix A reports the takes currently authorized for the fishery.

Coastal Migratory Pelagics Fishery

The CMP FMP was approved in 1982 and implemented by regulations effective in February of 1983. Managed species include king mackerel, Spanish mackerel, and cobia. The CMP FMP manages these species in federal waters in the Gulf of Mexico and in the Atlantic from Florida to New York. Spanish mackerel occur to depths of 75 m, cobia to depths of 125 m, and king mackerel to depths of 200 m. Consequently, fishing for CMP species typically occurs in waters less than 45 m but may occur in depths as great as 200 m. Fishing for CMP species in the Gulf of Mexico and Atlantic region is primarily conducted by hook-and-line, cast nets, and run-around and sink gillnets. Drift gillnets targeting CMP species have been prohibited since 1990, and many additional restrictions on gillnets targeting CMP were implemented in April 2000 via Amendment 9 to the CMP FMP.

Only the gillnet component of the authorized CMP fishery is known to adversely affect sea turtles. While sea turtles are typically vulnerable to capture on hooks, the hook-and-line gear used by both commercial and recreational fishers to target CMP species is limited to trolled or, to a much lesser degree (e.g., historically ~2% by landings for king mackerel), jigged handline, bandit, and rod-and-reel gear, i.e., techniques that are extremely unlikely to affect sea turtles (NMFS 2015).

A June 18, 2015 Opinion, as amended via a November 18, 2017 memorandum and attachment, comprises the most recent completed Section 7 consultation on the operation of the CMP fishery in the Gulf of Mexico and South Atlantic. The 2015 Opinion, as amended, concluded that the proposed action may adversely affect but is not likely to jeopardize the continued existence of any of the listed sea turtle species (i.e., green North Atlantic and South Atlantic DPS, hawksbill, Kemp's ridley, leatherback, and loggerhead NWA DPS). An ITS was provided, and Appendix A reports the takes currently authorized for the fishery.

Dolphin/Wahoo Fishery

The South Atlantic FMP for the dolphin/wahoo fishery was approved in December 2003. Under the Dolphin Wahoo FMP, dolphin and wahoo are managed from the east coast of Florida to Maine. The stated purpose of the Dolphin and Wahoo FMP is to adopt

precautionary management strategies to maintain the current harvest level and historical allocations of dolphin (90% recreational) and ensure no new fisheries develop. The FMP was developed when commercial dolphin landings in the Atlantic increased in the mid to late 1990's, due in part to an increasing number of longliners targeting dolphin or modifying their fishing practices such that dolphin and wahoo constitute a greater portion of their longline trips. At that time, HMS pelagic logline vessels were also fishing for dolphin using small hooks attached to their surface buoys and there were concerns regarding the potential for efforts shifts in the historical HMS longline fishery into more coastal waters (traditional recreational fishing grounds) to target dolphin because of increasing regulations and time and area closures for HMS. The commercial longline fishery for dolphin in the Atlantic consisted of approximately 3 or 4 longline vessels that direct effort on dolphin on a regular basis off the coasts of North and South Carolina (NMFS, 1995 & 1996) and longliners who catch dolphin and wahoo but primarily target HMS. NMFS conducted a formal Section 7 consultation that considered the effects on sea turtles of the proposed fishing actions that would be authorized under the FMP (NMFS 2003b). The August 27, 2003 Opinion concluded that green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles may be adversely affected by the longline component of the fishery, but it was not expected to jeopardize their continued existence. An ITS for sea turtles was provided with the Opinion. Pelagic longline vessels can no longer target dolphin/wahoo with smaller hooks because of hook size requirements in the HMS pelagic longline fishery, thus little longline effort targeting dolphin is currently believed to be present in the action area. Appendix A reports the takes currently authorized for the fishery.

Atlantic Sea Scallop Trawl and Dredge Fisheries

The Atlantic sea scallop fishery has a long history of operation in Mid-Atlantic, as well as New England waters (NEFMC 1982 ; NEFMC 2003). The fishery operates in areas and at times that it has traditionally operated and uses traditionally fished gear (NEFMC 1982 ; NEFMC 2003). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2007a). On Georges Bank and in the Mid-Atlantic, sea scallops are harvested primarily at depths of 30-100 m, while the bulk of landings from the Gulf of Maine are from relatively shallow nearshore waters (< 40 m) (NEFSC 2007a). Effort (in terms of days fished) in the Mid-Atlantic is about half of what it was prior to implementation of Amendment 4 to the Scallop FMP in the 1990s (NEFSC 2007a).

NMFS completed a Section 7 consultation on the Atlantic sea scallop fishery (NMFS 2008a). The Opinion concluded that the operation of the fishery was likely to adversely affect green, Kemp's ridley, leatherback, and loggerhead sea turtles, but was not likely to jeopardize their continued existence; an ITS was issued. Green, Kemp's ridley, and loggerhead sea turtles have been reported by NMFS-trained observers as being captured in scallop dredges and trawl gear. Methods used to detect any sea turtle interactions with scallop fishing gear (dredge or trawl gear) were insufficient prior to increased observation coverage in 2001, which now documents that this fishery results in many loggerhead mortalities on an annual basis.

Consultation was reinitiated to address the listing of 5 DPSs of Atlantic sturgeon in April 2012, as well as additional information available since the last Opinion on the fishery's effects on sea turtles. Reports by Murray (2011) and Warden and Murray (2011) provide new information on the annual number of sea turtle interactions in both the dredge and trawl components of the fishery. In addition, a workshop convened by NMFS to refine methods to determine the levels of serious injury/mortality to sea turtles interacting with Northeast fisheries, and papers by Milliken et al. (2007), Smolowitz et al. (2010) and the Scallop Plan Development Team, provided new information on the levels of serious injury/mortality to sea turtles in the fishery. Additionally, new management measures meant to reduce the impacts of the fishery on sea turtles were implemented since the completion of the last Opinion. The most recent consultation was completed in 2015 and the Opinion and Incidental Take Statement was issued on May 1, 2015. Appendix A reports the takes currently authorized for the Atlantic scallop trawl and dredge fisheries.

Northeast multispecies, monkfish, spiny dogfish, Atlantic bluefish, Northeast skate complex, Atlantic mackerel/squid/butterfish, and summer flounder/scup/black sea bass FMP Fisheries

In December 2013, NMFS completed the most recent Opinion on the effects of authorizing the (1) Northeast multispecies, (2) monkfish, (3) spiny dogfish, (4) Atlantic bluefish, (5) Northeast skate complex, (6) Atlantic mackerel/squid/butterfish, and (7) summer flounder/scup/black sea bass fisheries on sea turtles in a single “batched” consultation (i.e., NMFS 2013). Although these fisheries of the northeast and Mid-Atlantic regions are managed under 7 different FMPs, fishing activity under the different FMPs often occurs simultaneously and on the same vessel. Consequently, NMFS analyzed the effect of using various gear types across these fisheries due to the inability to attribute takes to individual FMPs. The consultation concluded that the operation of the fisheries, and the use of particular gears, were likely to adversely affect but not jeopardize the continued existence of any species of sea turtle. Appendix A reports the takes currently authorized for these collective fisheries by gear type (i.e., gillnet, bottom trawl, and trap/pot). The fisheries are described in the following paragraphs.

(1) Northeast Multispecies Fishery

The Northeast multispecies fishery operates throughout the year, with peaks in the spring and from October through February. Multiple gear types are used in the fishery including sink gillnet, trawl, and pot/trap gear, which are known to be a source of injury and mortality to right, humpback, and fin whales as well as loggerhead and leatherback sea turtles as a result of entanglement and capture in the gear (NMFS 2001a). The Northeast multispecies sink gillnet fishery has historically occurred from the periphery of the Gulf of Maine to Rhode Island in water as deep as 360 ft. 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. 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.

(2) Monkfish Fishery

The federal monkfish fishery occurs from Maine to the North Carolina/South Carolina border and is jointly managed by the New England Fishery Management Council (NEFMC) and Mid-Atlantic Fishery Management Council, under the Monkfish FMP (NMFS 2005b). Monkfish are harvested commercially primarily from the deeper waters of the Gulf of Maine, Georges Bank, and southern New England, and in the Mid-Atlantic. Monkfish have been found in depths ranging from the tide line to 900 m with concentrations between 70 and 100 m and at 190 m. The directed monkfish fishery uses several gear types that may entangle protected species, including gillnet and trawl gear.

Gillnet gear used in the monkfish fishery is known to capture ESA-listed sea turtles. Two unusually large stranding events occurred in April and May 2000 during which 280 sea turtles (275 loggerheads and 5 Kemp's ridleys) washed ashore on ocean facing beaches in North Carolina. Although there was not enough information to specifically determine the cause of the sea turtle deaths, there was information to suggest that the turtles died as a result of entanglement with large-mesh gillnet gear. The monkfish gillnet fishery, which uses a large-mesh gillnet, was known to be operating in waters off 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-in (20.3 cm) stretched mesh, in federal waters (3-200 nmi) off of North Carolina and Virginia. These restrictions were published in an Interim Final Rule under the authority of the ESA (67 FR 13098, March 21, 2002) and were implemented to reduce the impact of the monkfish and other large-mesh gillnet fisheries on endangered and threatened species of sea turtles in areas where sea turtles are known to concentrate. Following review of public comments submitted on the Interim Final Rule, NMFS published a Final Rule on December 3, 2002, that established the restrictions on an annual basis.

(3) Spiny Dogfish Fishery

The primary gear types for the spiny dogfish fishery are sink gillnets, otter trawls, bottom longline, and driftnet gear (NEFSC 2003). The predominance of any 1 gear type has varied over time (NEFSC 2003). In 2005, 62.1% of landings were taken by sink gillnet gear, followed by 18.4% in otter trawl gear, 2.3% in line gear, and 17.1% in gear defined as "other" (excludes drift gillnet gear) (NEFSC 2006). More recently, data from fish dealer reports in Fiscal Year 2008 indicate that spiny dogfish landings came mostly from sink gill nets (68.2%), and hook gear (15.2%), bottom otter trawls (4.9%), as well as unspecified (7.7%) or other gear (3.9%) (MAFMC 2010). Sea turtles can be incidentally captured in spiny dogfish gear, which can lead to injury and death as a result of forced submergence in the gear.

(4) Atlantic Bluefish Fishery

The fishery has been operating in the U.S. Atlantic (from Maine to Florida) for at least the last half century, although its popularity did not heighten until the late 1970s and early 1980s (MAFMC and ASMFC 1998). The majority of commercial fishing activity in the North Atlantic and Mid-Atlantic occurs in the late spring to early fall, when bluefish (and sea turtles) are most abundant in these areas (NEFSC 2005). This fishery is known to interact with loggerhead sea turtles, given the time and locations where the fishery occurs.

Gillnets account for the vast majority of bluefish landed by commercial harvesters. 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 harvest of bluefish include trawl, longline, handline, bandit, rod and reel, pot, trap, seine, and dredge gear (50 CFR 600.725(v)).

(5) Skate Fishery

The skate fishery has typically been composed of both a directed fishery and an indirect fishery. Otter trawls are the primary gear used to land skates in the U.S., with some landings also coming from sink gillnet, longline, and other gear (NEFSC 2007b). Bottom trawl gear accounted for 94.5% of directed skate landings. Gillnet gear is the next most common gear type, accounting for 3.5% of skate landings. All gears used to land skates are known to capture sea turtles.

(6) Mackerel/Squid/Butterfish Fisheries

Atlantic mackerel/squid/butterfish fisheries are managed under a single FMP, which was first implemented on April 1, 1983. Bottom otter trawl gear is the primary gear type used to land *Loligo* and *Illex* squid. Based on NMFS dealer reports, the majority of *Loligo* and *Illex* squid are fished in the Mid-Atlantic including waters within the action area of this consultation where loggerheads also occur. While squid landings occur year round, the majority of *Loligo* squid landings occur in the fall through winter months while the majority of *Illex* landings occur from June through October (MAFMC 2007a); time periods that overlap in whole or in part with the distribution of loggerhead sea turtles in Mid-Atlantic waters. Gillnets account for a small amount of landings in the mackerel fishery. 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.

(7) Summer Flounder, Scup, and Black Sea Bass Fisheries

In the Mid-Atlantic, summer flounder, scup, and black sea bass (BSB) are managed under a single FMP since these species occupy similar habitat and are often caught at the same time. Bottom otter and beam trawl gear are used most frequently in the commercial fisheries for all 3 species (MAFMC 2007b). Gillnets, handlines, dredges, and pots/traps are also occasionally used (MAFMC 2007b).

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 gear used in fisheries for other species like scup and BSB). TEDs are required throughout the year for trawl nets fished from the North Carolina/South Carolina border to Oregon Inlet, North Carolina, and seasonally (March 16-January 14) for trawl vessels fishing between Oregon Inlet, North Carolina, and Cape Charles, Virginia. Effort in the summer flounder, scup, and BSB fisheries has also declined since the 1980s and since each species 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 risk of sea turtle captures causing injury and death in summer flounder, scup, and BSB fishing gear.

Other Northeast and Mid-Atlantic Fisheries (American Lobster, and Red Crab)

Not all Northeast and Mid-Atlantic FMP-managed fisheries were included in the batched consultation. There are other Northeast and Mid-Atlantic fisheries federally managed under FMPs, the effects of which have been consulted on separately. Consultations on these fisheries have concluded each is not likely to jeopardize listed sea turtles, with anticipated annual take levels. Each has been the subject of non-jeopardy conclusions and have low anticipated incidental take levels, which are reported in Appendix A.

4.2.1.2 Fisheries Independent Monitoring

NMFS Integrated Fisheries Independent Monitoring Activities in the Southeast Region promotes and funds projects conducted by the SEFSC and other NMFS partners to collect fisheries independent data. The various projects use a variety of gear (e.g., trawls, nets, etc.) to conduct fishery research. Sea turtles are incidentally taken during the course of these activities. An Opinion was issued in May 2016, concluding the activities are likely to adversely affect, but not likely to jeopardize, the continued existence of any sea turtle species. Up to 34 loggerhead, 22 Kemp's ridley, 1 leatherback, and 18 green sea turtle lethal takes are expected over continuing 5 year periods and authorized in the ITS (NMFS 2016).

In June 2016, NMFS completed a Programmatic Consultation on the Continued Prosecution of Fisheries and Ecosystem Research Conducted and Funded by the Northeast Fisheries Science Center, concluding the activities are likely to adversely affect, but not likely to jeopardize, the continued existence of any sea turtle species. Sea turtles are incidentally taken during the course of these activities. Up to 10 loggerhead, 15 Kemp's ridley, and 5 leatherback lethal takes are expected over continuing 5 year periods and authorized in the ITS.

In January 2017, NMFS completed a consultation on USFWS funding of the Georgia Department of Natural Resources Coastal Resources Division (GCRD) to collect, analyze and report biological and fisheries information to describe the conditions or health of recreationally important finfish populations and develop management recommendations that would maintain or restore the stocks in coastal Georgia. The Opinion concludes the activities are likely to adversely affect, but are not likely to jeopardize the continued existence of, any sea turtle species. Up to 1 loggerhead, 2 Kemp's ridley, 2 North Atlantic DPS of green sea turtle, and 1 South Atlantic DPS of green sea turtle are expected over continuing 5 year periods and authorized in the ITS.

4.2.1.3 ESA Section 10 Scientific Research Permits

The ESA allows for the issuance of permits authorizing take of certain ESA-listed species for the purposes of scientific research or enhancement (Section 10(a)(1)(A)). NMFS

consults with itself to ensure that issuance of such permits can be done in compliance with Section 7 of the ESA.

Sea turtles are the focus of research activities in the action area for which take is authorized by Section 10 permits under the ESA. There were 31 active scientific research permits directed toward sea turtles that are applicable to the action area of this Opinion.

Authorized activities range from photographing, weighing, and tagging sea turtles, to blood sampling, tissue sampling (biopsy), and performing laparoscopy. The number of authorized takes varies widely depending on the research and species involved but may involve the taking of hundreds of sea turtles annually. Most takes authorized under these permits are expected to be nonlethal. Before any research permit is issued, the proposal must be reviewed under the permit regulations (i.e., must show a benefit to the species). In addition, since issuance of the permit is a federal activity, Section 7 analysis is also required to ensure the issuance of the permit is not likely to result in jeopardy to the species. Permits are issued for 5 years.

4.2.1.4 Dredging

Marine dredging vessels are common within U.S. coastal waters. Although the underwater noises from dredge vessels are typically continuous in duration (for periods of days or weeks at a time) and strongest at low frequencies, they are not believed to have any long-term effect on sea turtles. However, the construction and maintenance of federal navigation channels, expansion of harbors, dredging in sand mining sites ("borrow areas"), and some beach nourishment activities, have been identified as sources of sea turtle mortality. Hopper dredges in the dredging mode are capable of moving relatively quickly compared to sea turtle swimming speed and can thus overtake, entrain, and kill sea turtles as the suction draghead(s) of the advancing dredge overtakes the resting or swimming turtle. Entrained sea turtles rarely survive. NMFS completed a regional Opinion on the impacts of USACE's hopper-dredging in the South Atlantic in 1997 (NMFS 1997b). NMFS determined that (1) hopper dredging in the South Atlantic would adversely affect shortnose sturgeon and 4 sea turtle species (i.e., green, hawksbill, Kemp's ridley, and loggerheads), but would not jeopardize their continued existence, and (2) South Atlantic dredging would not adversely affect leatherback sea turtles or ESA-listed large whales. An ITS for those species adversely affected was issued. The USACE requested reinitiation of consultation in 2007 to: (1) consider species and critical habitat, that may be affected by the action, which had not been listed at the time of the previous Opinion and were not considered (e.g., smalltooth sawfish, ESA-listed corals, *Acropora* critical habitat); (2) update the areas, channels, and dredge techniques that the USACE wanted considered, and (3) to include BOEM as a co-action agency. NMFS is currently working on drafting an Opinion.

4.2.1.5 Federal Military Activities

Potential sources of adverse effects in the action area include operations of the U.S. DoD. The U.S. Navy (USN) conducts military readiness activities, which can be categorized as either training or testing exercises, throughout the action area. During training, existing

and established weapon systems and tactics are used in realistic situations to simulate and prepare for combat. Activities include: routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. USN performs testing activities to ensure that its military forces have the latest technologies and techniques available to them. USN activities are likely to produce noise and harass sea turtles throughout the action area. Formal consultations on overall USN activities in the Atlantic have been completed, including USN Joint Logistics Over-the-Shore Training in Virginia and North Carolina (JLOTS) 2014, [Opinion issued to USN in 2014 (NMFS 2014)]; USN Atlantic Fleet Training and Testing (AFTT) Activities (2013-2018), [Opinion issued to USN in 2013 (NMFS 2013)]; U.S. Navy East Coast Range Complex, [Opinion issued to USN in 2012 (NMFS 2012)]; USN's Activities in East Coast Training Ranges [Opinion issued to USN in 2011 (NMFS June 1, 2011)]; USN Atlantic Fleet Sonar Training Activities (AFAST) [Opinion issued to USN in 2011 (January 20, 2011)]; Navy AFAST LOA 2012-2014: U.S. Navy active sonar training along the Atlantic Coast and Gulf of Mexico [Opinion issued to USN in 2011 (December 19, 2011)]; and Navy's East Coast Training Ranges (Virginia Capes, Cherry Point, and Jacksonville) [Opinion issued to USN in 2010 (June 2010)]. These Opinions concluded that although there is a potential from some USN activities to affect sea turtles, those effects were not expected to impact any species on a population level. Therefore, the activities were determined to be not likely to jeopardize the continued existence of any ESA-listed sea turtle species, or destroy or adversely modify critical habitat of any listed species.

4.2.2 State or Private Actions

4.2.2.1 Private and Commercial Vessel Operations

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. Commercial traffic and recreational pursuits can also adversely affect sea turtles through propeller- and boat strikes. The STSSN includes many records of vessel interaction (propeller injury) with sea turtles off south Atlantic coastal states such as Florida, where there are high levels of vessel traffic. The extent of the problem is difficult to assess because of not knowing whether the majority of sea turtles are struck pre- or post-mortem. Private vessels in the action area participating in high-speed marine events (e.g., boat races) are a particular threat to sea turtles. It is important to note that although minor vessel collisions may not kill an animal directly, they may weaken or otherwise affect an animal, which makes it more likely to become vulnerable to effects such as entanglements.

4.2.3 Climate Change

As discussed earlier in this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. Potential effects commonly mentioned include changes in sea temperatures and salinity (due to melting ice and

increased rainfall), ocean currents, storm frequency and weather patterns, and ocean acidification. These changes have the potential to affect species behavior and ecology including migration, foraging, reproduction (e.g., success), and distribution. For example, sea turtles currently range from temperate to tropical waters. A change in water temperature could result in a shift or modification of range. Climate change may also affect marine forage species, either negatively or positively (the exact effects for the marine food web upon which sea turtles rely is unclear, and may vary between species). It may also affect migratory behavior (e.g., timing, length of stay at certain locations). These types of changes could have implications for sea turtle recovery.

Additional discussion of climate change can be found in the Status of the Species section (Section 3.1). However, to summarize with regards to the action area, global climate change may affect the timing and extent of population movements and their range, distribution, species composition of prey, and the range and abundance of competitors and predators. 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 are all possible impacts that may occur as the result of climate change. Still, more information is needed to better determine the full and entire suite of impacts of climate change on sea turtles and specific predictions regarding impacts in the action area are not currently possible.

4.2.4 Marine Pollution

While some sources of marine pollution are difficult to attribute to a specific federal, state, local or private action, they may indirectly affect sea turtles in the action area. Sources of pollutants include atmospheric loading of pollutants such as PCBs and stormwater runoff from coastal towns and cities into rivers and canals emptying into bays and the ocean (e.g., Mississippi River). There are studies on organic contaminants and trace metal accumulation in green, leatherback, and loggerhead sea turtles (Aguirre et al. 1994; Caurant et al. 1999; Corsolini et al. 2000). McKenzie et al. (1999) measured concentrations of chlorobiphenyls and organochlorine pesticides in sea turtle tissues collected from the Mediterranean (Cyprus, Greece) and European Atlantic waters (Scotland) between 1994 and 1996. Omnivorous loggerhead turtles had the highest organochlorine contaminant concentrations in all the tissues sampled, including those from green and leatherback turtles (Storelli et al. 2008b). It is thought that dietary preferences were likely to be the main differentiating factor among species. Decreasing lipid contaminant burdens with sea turtle size were observed in green turtles, most likely attributable to a change in diet with age. (Sakai et al. 1995) documented the presence of metal residues occurring in loggerhead sea turtle organs and eggs. Storelli et al. (1998) analyzed tissues from 12 loggerhead sea turtles stranded along the Adriatic Sea (Italy) and found that characteristically, mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991a). No information on detrimental threshold concentrations is available and little is known about the consequences of exposure of organochlorine compounds to sea turtles. Research is needed into how chlorobiphenyl, organochlorine, and heavy-metal accumulation effect the short- and long-term health of sea

turtles and what effect those chemicals have on the number of eggs laid by females. More information is needed to understand the potential impacts of marine pollution in the action area.

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, stimulate plankton blooms in closed or semi-closed estuarine systems. Oxygen depletion, referred to as hypoxia, can negatively impact sea turtles' habitats, prey availability, and survival and reproductive fitness. But the effects of nutrient loading on larger embayments (and the pelagic environment of the action area) are unknown.

Fuel oil spills could affect animals directly or indirectly through the food chain. Fuel spills involving fishing vessels are common events, although these spills typically involve small amounts of material. Larger oil spills may result from accidents, although these events would be rare. No direct adverse effects on listed species resulting from fishing vessel fuel spills have been documented.

4.2.5 Conservation and Recovery Actions Benefiting Sea Turtles in the Action Area

NMFS has implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for the Atlantic HMS pelagic longline and for other hook-and-line fisheries (i.e., Gulf of Mexico reef fish and South Atlantic snapper-grouper permitted hook-and-line fisheries), TED requirements for the Southeast shrimp trawl and North Carolina flynet fisheries, mesh size restrictions in the North Carolina gillnet fishery and Virginia's gillnet fisheries, and area closures in the North Carolina gillnet fishery. In addition to regulations, outreach programs have been established and data on sea turtle interactions with recreational fisheries has been collected through the Marine Recreational Fishery Statistical Survey (MRFSS)/Marine Recreational Information Program. The summaries below discuss all of these measures in more detail.

Reducing Threats from Pelagic Longline and Other Hook-and-Line Fisheries

On July 6, 2004, following consultation on the effects of the HMS pelagic longline fishery, NMFS published a Final Rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic pelagic longline fishery (69 FR 40734). The management measures include mandatory circle hook and bait requirements, mandatory attendance of vessel owners and operators at Safe Handling, Release, and Identification workshops, and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality.

NMFS published Final Rules to implement sea turtle release gear requirements and sea turtle careful release protocols in the South Atlantic snapper-grouper fishery (November 8, 2011, 76 FR 69230). These measures require owners and operators of vessels with federal commercial or charter vessel/headboat permits for South Atlantic snapper-grouper to comply with sea turtle (and smalltooth sawfish) release protocols and have on board specific sea turtle-release gear.

Revised Use of Turtle Excluder Devices in Trawl Fisheries

NMFS has also implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles in commercial shrimp trawl fisheries. In particular, NMFS has required the use of TEDs in southeast U.S. shrimp trawls since 1989 and in summer flounder trawls in the Mid-Atlantic area (south of Cape Charles, Virginia) since 1992. It has been estimated that TEDs exclude 97% of the sea turtles caught in such trawls. These regulations have been refined over the years to ensure that TED effectiveness is maximized through more widespread use, and proper placement, installation, configuration (e.g., width of bar spacing), and floatation.

Significant measures have been developed to reduce sea turtle interactions in summer flounder trawls and trawls that meet the definition of a summer flounder trawl (which would include fisheries for other species like scup and BSB) by requiring TEDs in trawl nets fished from the North Carolina/South Carolina border to Cape Charles, Virginia. However, the TED requirements for the summer flounder trawl fishery do not require the use of the larger TEDs that are used in the shrimp trawl fisheries to exclude leatherbacks, as well as large benthic-immature and sexually mature loggerheads and green sea turtles.

In 1998, the SEFSC began developing a TED for flynets. In 2007, the Flexible Flatbar Flynet TED was developed and catch retention trials and usability testing was completed (Gearhart 2010). Experiments are still ongoing to certify a bottom-opening flynet TED.

Placement of Fisheries Observers to Monitor Sea Turtle Captures

On August 3, 2007, NMFS published a Final Rule that required selected fishing vessels to carry observers on board to collect data on sea turtle interactions with fishing operations, to evaluate existing measures to reduce sea turtle captures, and to determine whether additional measures to address prohibited sea turtle captures may be necessary (72 FR 43176). This Rule also extended the number of days NMFS observers could be placed aboard vessels, for 30-180 days, in response to a determination by the Assistant Administrator that the unauthorized take of sea turtles may be likely to jeopardize their continued existence under existing regulations.

Final Rules for Large-Mesh Gillnets

In March 2002, NMFS published new restrictions for the use of gillnets with larger than 8-in-stretched mesh, in federal waters (3-200 nmi) off North Carolina and Virginia. These restrictions were published in an interim Final Rule under the authority of the ESA (67 FR 13098) and were implemented to reduce the impact of the monkfish and other large-mesh gillnet fisheries on ESA-listed sea turtles in areas where sea turtles are known to concentrate. Following review of public comments submitted on the interim Final Rule, NMFS published a Final Rule on December 3, 2002, that established the restrictions on an annual basis. As a result, gillnets with larger than 8-in-stretched mesh were not allowed in federal waters (3-200 nmi) in the areas described as follows: (1) north of the North Carolina/South Carolina border at the coast to Oregon Inlet at all times; (2) north of Oregon Inlet to Currituck Beach Light, North Carolina, from March 16-January 14; (3) north of Currituck Beach Light, North Carolina, to Wachapreague Inlet, Virginia, from April 1-January 14; and (4) north of Wachapreague Inlet, Virginia, to Chincoteague,

Virginia, from April 16-January 14. On April 26, 2006, NMFS published a Final Rule (71 FR 24776) that included modifications to the large-mesh gillnet restrictions. The new Final Rule revised the gillnet restrictions to apply to stretched mesh that is greater than or equal to 7 inches. Federal waters north of Chincoteague, Virginia, remain unaffected by the large-mesh gillnet restrictions.

Sea Turtle Handling and Resuscitation Techniques

NMFS published a Final Rule (66 FR 67495, December 31, 2001) detailing handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities. Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the Final Rule. These measures help to prevent mortality of hardshell turtles caught in fishing or scientific research gear.

Outreach and Education, Sea Turtle Rescue and Rehabilitation

There is an extensive network of SSTSSN participants along the Atlantic coast who not only collect data on dead sea turtles, but also rescue and rehabilitate any live stranded sea turtles.

A Final Rule (70 FR 42508) published on July 25, 2005, allows any agent or employee of NMFS, the USFWS, the 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, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. NMFS already affords the same protection to sea turtles listed as threatened under the ESA [50 CFR 223.206(b)].

4.3 Factors Affecting Smalltooth Sawfish within the Action Area

The following analysis examines actions that may affect this species and its environment specifically within the action area. The activities that shape the environmental baseline in the action area of this consultation are primarily federal fisheries. Other environmental impacts include effects of permits allowing take under the ESA and marine pollution.

4.3.1 Federal Actions

In recent years, NMFS has undertaken Section 7 consultations to address the effects of federally permitted fisheries and other federal actions on smalltooth sawfish, and when appropriate, has authorized the incidental taking of this species. Each of those consultations sought to minimize the adverse impacts of the action on smalltooth sawfish. The following sections summarize anticipated sources of incidental take of smalltooth sawfish in the action area that have already concluded formal Section 7 consultation or are currently undergoing formal Section 7 consultation or that have undergone early section 7 consultation.

4.3.1.1 Federal Fisheries

HMS Shark Fisheries

The current consultation considers the effects of the operation of HMS shark fisheries on smalltooth sawfish, thus effects associated with those activities are not considered part of the environmental baseline. However, the past operation of these fisheries, and their impacts are part of the environmental baseline for sawfish in the action area.

Section 2 of this Opinion provides an overview of the history of HMS shark fisheries and their management. These fisheries include commercial shark bottom longline and gillnet fisheries and recreational shark fisheries. The consultation history on these fisheries is described in Section 1. Both HMS shark bottom longlines and shark gillnets set within the range of smalltooth sawfish have resulted in adverse effects on smalltooth sawfish in the past, with the majority of adverse effects attributed to bottom longlines, particularly in the vicinity of the Florida Keys and Dry Tortugas.

Appendix A reports the anticipated smalltooth sawfish incidental takes for this fishery prior to this consultation, per the 2012 Opinion.

Southeastern U.S. Shrimp Fishery

Smalltooth sawfish were historically caught as bycatch in otter trawls (NMFS 2000). Early literature accounts document smalltooth sawfish as being frequently caught by shrimp trawls. For example, Bigelow and Schroeder (1953) noted smalltooth sawfish were of “considerable concern to fishermen as nuisances because of the damage they do to drift- and turtle-nets, to seines, and to shrimp trawls in which they often become entangled; and because of the difficulty of disentangling them without being injured by their saws.” Entangled smalltooth sawfish frequently had to be cut free, causing extensive damage to trawl nets and presenting a substantial hazard if brought on board. Most smalltooth sawfish caught by fishermen were either killed outright or released only after removal of their saw. The smalltooth sawfish recovery plan (NMFS 2009d) states that available data on interactions between trawl fisheries and the U.S. DPS of smalltooth sawfish are very limited, but that shrimp trawl fisheries are associated with high sawfish mortality per interaction.

Since the species was listed in 2003, reports of smalltooth sawfish capture in shrimp trawls have been rare. NMFS completed its first Section 7 consultations on the impacts to smalltooth sawfish from the shrimp fishery in the Gulf of Mexico (NMFS 2006) and the South Atlantic (NMFS 2005a) in 2005 and 2006, respectively. These consultations found these fisheries likely to adversely affect smalltooth sawfish, but not likely to jeopardize its continued existence. The ITSs provided for these Opinions each anticipated the lethal take of 1 smalltooth sawfish annually in the fishery based on anecdotal reports. However, in May 2009 and March 2010, NMFS requested reinitiation of Section 7 consultation on the South Atlantic shrimp fishery and Gulf of Mexico shrimp fishery, respectively to analyze their effects on smalltooth sawfish because new observer data indicated that the incidental take statements of the respective Opinions had been exceeded. In July 2007, NMFS implemented a mandatory observer program component for the Gulf of Mexico federal

shrimp fishery. Similarly, in 2008 a mandatory observer program was initiated for the South Atlantic federal shrimp fishery. With coverage levels in these fisheries of about 2% of total effort via these mandatory programs, NMFS documented 3 captures in 2009 and 2 in 2010. On May 9, 2012, NMFS completed the new Opinion, which analyzed the implementation of the Southeast U.S. shrimp fisheries in federal waters under the Magnuson-Stevens Fisheries Conservation and Management Act. The Opinion also considered a proposed amendment to the sea turtle conservation regulations that would withdraw the alternative tow time restriction at 50 CFR 223.206(d)(2)(ii)(A)(3) for skimmer trawls, pusher-head trawls, and wing nets (butterfly trawls) and instead require all of these vessels to use TEDs. The Opinion concluded that the proposed action was not likely to jeopardize the continued existence of smalltooth sawfish. An ITS was provided.

On November 21, 2012, NMFS determined that a Final Rule requiring TEDs in skimmer trawls, pusher-head trawls, and wing nets was not warranted and withdrew the proposal. NMFS reinitiated consultation on November 26, 2012 because of the decision to not implement the Final Rule created a change to the proposed action analyzed in the 2012 Opinion. The updated opinion was completed in April 2014, and again NMFS determined the operation of the southeast U.S. shrimp fisheries in federal waters was not likely to jeopardize the continued existence smalltooth sawfish. Appendix A reports the smalltooth sawfish incidental takes authorized for this fishery under the 2014 ITS.

Coastal Migratory Pelagic Fishery

NMFS completed a Section 7 consultation on the operation of the coastal migratory pelagic fishery in the Atlantic and Gulf of Mexico (NMFS 2015b). Gillnets are the primary gear type used by commercial fishers in the South Atlantic, while the recreational sector uses hook-and-line gear. The Opinion concluded that smalltooth sawfish may be adversely affected by operation of the fishery; however, the proposed action was not expected to jeopardize its continued existence, and an ITS was provided. Appendix A reports the smalltooth sawfish incidental takes authorized for this fishery.

Gulf of Mexico Reef Fish Fishery

The Gulf of Mexico reef fish fishery used to use three basic types of gear: spear or powerhead, trap, and hook-and-line gear. Hook-and-line gear used in the fishery includes both commercial bottom longline and commercial and recreational vertical line (handline, bandit gear, rod and reel). Trap gear was phased-out completely by February 2007, but prior to that likely resulted in a few smalltooth sawfish entanglements. The hook-and-line components of the fishery have likely always had the most adverse effects on smalltooth sawfish. However, all consultations to date have concluded the fishery is not likely to jeopardize the continued existence of the smalltooth sawfish. The most recent opinion was issued on September 30, 2011. An ITS was provided authorizing a small number of non-lethal takes in the commercial and recreational hook-and-line components of the fishery.

South Atlantic Snapper-Grouper Fishery

The fishery uses spear and powerheads, black sea bass pot, and hook-and-line gear. Hook-and-line gear used in the fishery includes commercial bottom longline gear and commercial and recreational vertical line gear (e.g., handline, bandit gear, and rod-and-reel). The most

recent opinion on the fishery was completed in 2016 and concluded the proposed action was not likely to jeopardize the continued existence of any smalltooth sawfish; an ITS was issued. Appendix A reports the takes authorized for this fishery.

Gulf of Mexico and South Atlantic Spiny Lobster Fishery

NMFS completed a Section 7 consultation on the Gulf and South Atlantic Spiny Lobster FMP on August 27, 2009 (NMFS 2009d). The commercial component of the fishery consists of diving, bully net and trapping sectors; recreational fishers are authorized to use bully net, and hand-harvest gears. Of the gears used, traps are expected to result in adverse effects on smalltooth sawfish. The consultation determined the operation of the fishery would not jeopardize any listed species. An ITS was issued for takes in the commercial trap sector of the fishery.

4.3.1.2 ESA Section 10 Scientific Research Permits

Section 10(a)(1)(A) of the ESA allows NMFS to issue permits for the taking of ESA-listed species for scientific research or enhancement purposes. NMFS consults with itself to ensure that its issuance of these permits can be done in compliance with Section 7 of the ESA. There are currently 4 active research permits issued for smalltooth sawfish. The permits allow researchers to capture, handle, collect tissue and blood samples, and tag smalltooth sawfish. Although the research may result in disturbance and minor injury of smalltooth sawfish, the activities are not expected to affect the reproduction of the individuals that are caught, nor result in mortality.

4.3.2 State or Private Actions

Entanglement in state trap/pot fisheries is another potential route of effect to smalltooth sawfish. The State of Florida's stone crab fishery is an example of a state trap fishery that may interact with smalltooth sawfish. On October 15, 2011, NMFS repealed the federal FMP for stone crab. Prior to the repeal, NMFS prepared an opinion on the operation of the federal fishery. The Opinion concluded the federal stone crab fishery was likely to adversely affect smalltooth sawfish, but it was not likely to jeopardize their continued existence. The State of Florida now exclusively manages the stone crab fishery, even vessels fishing in the EEZ (which includes the action area). The State of Florida has actively managed the fishery since 1929; the federal FMP was implemented in 1979 to address gear conflicts. The federal fishery was managed primarily by issuing regulations complimentary to those promulgated by the State of Florida. Since the State of Florida has essentially been the lead management agency for the state and federal fishery for some time, little change in how the fishery operates or amount of the effort occurring in the fishery is expected because of the repeal of the federal FMP. Therefore, the anticipated adverse effects described in the Opinion completed before the repeal of the federal FMP are expected to continue to occur to smalltooth sawfish.

Additionally, lost fishing gear, or discarded hooks and line, can also pose an entanglement threat to smalltooth sawfish.

4.3.3 Climate Change

As discussed earlier in this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. Potential effects to the environment include changes in sea temperatures and salinity (due to melting ice and increased rainfall), ocean currents, storm frequency and weather patterns, and ocean acidification. These changes have the potential to affect species behavior and ecology including migration, foraging, reproduction (e.g., success), and distribution.

Additional discussion of climate change can be found in the Status of the Species section (Section 3.3.7). However, more information is needed to better determine the full and entire suite of impacts of climate change on this species and specific predictions regarding impacts in the action area are not currently possible.

4.3.4 Other Potential Sources of Impacts in the Environmental Baseline

Marine Pollution

Smalltooth sawfish have been encountered with polyvinyl pipes and fishing gear entangled on their toothed rostrum (Seitz and Poulakis 2006). The same sources of pollutants described previously for sea turtles (see Section 4.3.4) may also adversely affect smalltooth sawfish.

4.3.5 Conservation and Recovery Actions Shaping the Environmental Baseline

Regulations restricting the use of gear known to incidentally catch smalltooth sawfish may benefit the species by reducing their incidental capture and/or mortality in fishing gear. In 1994, entangling nets (including gillnets, trammel nets, and purse seines) were banned in Florida state waters. Although intended to restore the populations of inshore gamefish, this action removed possibly the greatest source of fishing mortality on smalltooth sawfish (Simpfendorfer 2002).

Public Outreach

Public outreach efforts are also helping to educate the public on smalltooth sawfish status and proper handling techniques and helping to minimize interaction, injury, and mortality of encountered smalltooth sawfish. Information regarding the status of smalltooth sawfish and what the public can do to help the species is available on the websites of the Florida Museum of Natural History,¹⁹ NMFS,²⁰ and the Ocean Conservancy.²¹ Reliable information is also available at websites maintained by noted sawfish expert Matthew McDavitt.²² These organizations and individuals also educate the public about sawfish status and conservation through regular presentations at various public meetings.

Smalltooth Sawfish Recovery Plan

¹⁹ <http://www.flmnh.ufl.edu/fish/Sharks/Sawfish/SRT/srt.htm>

²⁰ <http://www.sero.nmfs.noaa.gov/pr/SmalltoothSawfish.htm>

²¹ http://www.oceanconservancy.org/site/PageServer?pagename=fw_sawfish

²² <http://hometown.aol.com/nokogiri/>

In September 2003, NMFS convened a smalltooth sawfish recovery team. Under Section 4(f)(1) of the ESA, NMFS is required to develop and implement recovery plans for the conservation and survival of endangered and threatened species. The final smalltooth sawfish recovery plan published on January 21, 2009 (74 FR 3566). The recovery plan is available at <http://sero.nmfs.noaa.gov/pr/SmalltoothSawfish.htm>.

4.4 Factors Affecting Atlantic Sturgeon within the Action Area

The five DPSs of Atlantic sturgeon on the East Coast of the U.S. mix extensively in marine waters (Erickson et al. 2011; Stein et al. 2004c). During various seasons and portions of their life cycles, individual fish will make migrations into rivers, nearshore waters, and offshore waters in the North Atlantic Ocean. Adult and sub-adult (age 2 fish or older) spend a considerable portion of their lives in coastal and marine waters (ASSRT 2007; Collins and Smith 1997; Laney et al. 2007; Munro et al. 2007; Stein et al. 2004c) where they are subject to bycatch mortality by commercial fisheries (Armstrong and Hightower 2002; Collins et al. 1996; Spear 2007; Stein et al. 2004a; Trencia et al. 2002), poor water quality in certain estuaries (Collins et al. 2000a; Dadswell 2006) and other potential threats, such as dredging, and alteration of spawning and foraging habitat (ASSRT 2007; Munro et al. 2007). Because the action area encompasses the entire marine range of Atlantic sturgeon in the U.S., the statuses of the five DPSs presented in Section 3.0 of this Opinion most accurately reflect the species' statuses within the action area. Likewise, while the following discussion of factors affecting the species reflects conditions both inside and outside of the immediate action area, this discussion most accurately reflects those factors acting on Atlantic sturgeon that may occur with the action area.

The following analysis examines actions that may affect the species and their environment specifically within the action area. The activities that shape the environmental baseline in the action area of this consultation are primarily federal fisheries. Other environmental impacts include effects of dredging, research, marine pollution and debris, and acoustic impacts.

4.4.1 Federal Actions

NMFS issues federal permits for a number of fisheries and other federal actions, and has undertaken a number of Section 7 consultations to address the effects of those activities on other threatened and endangered species, such as sea turtles. The summary below of federal actions and the effects these actions have had on Atlantic sturgeon includes only those federal actions in the action area that have already concluded or are currently undergoing formal Section 7 consultation or that have undergone early section 7 consultation.

4.4.1.1 Federal Fisheries

Atlantic sturgeon are adversely affected by fishing gears used throughout the action area. While a number of different gears are utilized (e.g., gillnet, longline, other types of hook-and-line gear, trawl gear, and pot fisheries), Atlantic sturgeon bycatch mainly occurs in gillnets, with the greatest number of captures and highest mortality rates occurring in sink

gillnets. Atlantic sturgeon are also taken in trawl fisheries, though recorded captures and mortality rates are low. For all fisheries for which there is an FMP or for which any federal action is taken to manage that fishery, impacts to listed species have been evaluated under Section 7.

Atlantic Sea Scallop

The Atlantic sea scallop fishery has a long history of operation in Mid-Atlantic, as well as New England waters (NEFMC 1982; NEFMC 2003). The fishery operates in areas and at times that it has traditionally operated and uses traditionally fished gear (NEFMC 1982; NEFMC 2003). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2007a). On Georges Bank and in the Mid-Atlantic, sea scallops are harvested primarily at depths of 30-100 m, while the bulk of landings from the Gulf of Maine are from relatively shallow nearshore waters (< 40 m) (NEFSC 2007a). Effort (in terms of days fished) in the Mid-Atlantic is about half of what it was prior to implementation of Amendment 4 to the Scallop FMP in the 1990s (NEFSC 2007a).

The most recent consultation was completed in 2012, and later amended in 2015 and 2018. The opinion found the fishery was not likely to jeopardize Atlantic sturgeon DPSs. Appendix A reports the takes currently authorized for the Atlantic scallop trawl and dredge fisheries.

Coastal Migratory Pelagic Resources Fisheries

NMFS completed a Section 7 consultation on the operation of the coastal migratory pelagic resources fishery in the Gulf of Mexico and South Atlantic (NMFS 2015b). In the Gulf of Mexico and South Atlantic, commercial fishers target king and Spanish mackerel with hook-and-line (i.e., handline, rod-and-reel, and bandit), gillnet, and cast net gears. Recreational fishers in both areas use only rod-and-reel. Trolling is the most common hook-and-line fishing technique used by both commercial and recreational fishers. Although run-around gillnets accounted for the majority of the king mackerel catch from the late 1950s through 1982, in 1986, and in 1993, handline gear has been the predominant gear used in the commercial king mackerel fishery since 1993 (NMFS 2015b). The consultation concluded that the operation of the coastal migratory pelagic resources fishery in the Gulf of Mexico and South Atlantic was likely to adversely affect, but not jeopardize, the continued existence of any DPS of Atlantic sturgeon. Incidental take was authorized and is reported in Appendix A.

HMS Atlantic Shark Fisheries

These fisheries are one of the subjects of this consultation. These fisheries include commercial shark bottom longline and gillnet fisheries and recreational shark fisheries. NMFS (2012c) was the first formal consultation that evaluated the potential adverse effects of these fisheries on Atlantic sturgeon DPSs. Hook-and-line gear (including bottom longline gear) is considered not likely to adversely affect Atlantic sturgeon. NMFS (2012c) considered the potential adverse effects from bottom longline gear on Atlantic sturgeon to be extremely unlikely to occur. It did, however, anticipate the capture of Atlantic sturgeon in shark and smoothhound gillnet gear, but it ultimately concluded the proposed action was not likely to jeopardize the continued existence of Atlantic sturgeon

and an ITS for the incidental take of Atlantic sturgeon by DPS was issued. Appendix A reports the takes authorized for the fishery prior to completion of this consultation.

Southeastern Shrimp Trawl Fisheries

On May 9, 2012, NMFS completed a new Opinion on the southeastern shrimp fisheries, which included an evaluation of the potential impacts of the fisheries on Atlantic sturgeon DPSs for the first time since they were listed. Information considered in the Opinion included the North Carolina Division of Marine Fisheries reporting that no Atlantic sturgeon were observed in 958 observed tows conducted by commercial shrimp trawlers working in North Carolina waters (L. Daniel, NCDMF, pers. comm., via public comment on the proposed rule to list Atlantic sturgeon, 2010). Nine Atlantic sturgeon have been reported captured in the South Atlantic shrimp trawl fisheries, seven Atlantic sturgeon were captured by a single shrimp trawler off Winyah Bay, South Carolina, from October 27-29, 2008). Of the latter, six were caught in the main otter trawl gear and 1 was captured in the try net. Six were released alive, 1 was released dead (NMFS 2014c). One Atlantic sturgeon was captured by a shrimp trawler off South Carolina near Kiawah Island, South Carolina, on December 13, 2011, and it was released alive. Two Atlantic sturgeon were captured by a shrimp trawler near Sapelo Island, Georgia, from December 27-29, 2011. Both were approximately 2 ft long, and both were released alive. No Atlantic sturgeon have been observed caught since 2011 (NMFS 2014c). Collins et al. (1996) did a study of commercial bycatch of shortnose and Atlantic sturgeon. Based on this and additional information, the 2012 Opinion, concluded that interactions between shrimp trawls and Atlantic sturgeon were likely but many of the animals were likely to survive the interactions. Ultimately, the Opinion concluded that the proposed action was likely to adversely affect Atlantic sturgeon, but was not likely to jeopardize the continued existence of any Atlantic sturgeon DPS. The most recent Opinion, completed in 2014, on southeastern shrimp trawl fisheries came to this same conclusion. The amount of incidental take authorized is reported in Appendix A.

Northeast multispecies, monkfish, spiny dogfish, Atlantic bluefish, Northeast skate complex, Atlantic mackerel/squid/butterfish, and summer flounder/scup/black sea bass FMP Fisheries

In December 2013, NMFS completed the most recent Biological Opinion on the effects of authorizing the (1) Northeast multispecies, (2) monkfish, (3) spiny dogfish, (4) Atlantic bluefish, (5) Northeast skate complex, (6) Atlantic mackerel/squid/butterfish, and (7) summer flounder/scup/black sea bass fisheries on Atlantic sturgeon in a “batched” consultation (see 4.2.1.1 for details regarding the scope of this 2013 opinion). The consultation concluded that the operation of the fisheries was likely to adversely affect but not jeopardize the continued existence of any DPS of Atlantic sturgeon; incidental take was authorized. Appendix A reports the Atlantic sturgeon DPS incidental takes authorized for these fisheries, by gear type associated with these fisheries (i.e., gillnet and bottom trawl). The fisheries included in the batched consultation are described below.

(1) Northeast Multispecies Fishery

The fishery includes the following species: American plaice, Atlantic cod, Atlantic halibut, Atlantic wolffish, haddock, ocean pout, offshore hake, pollock, redfish, red hake, silver hake, white hake, windowpane flounder, winter flounder, witch flounder, and yellowtail flounder. The Northeast multispecies sink gillnet fishery has historically occurred from the periphery of the Gulf of Maine to Rhode Island in water as deep as 360 ft. 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 because extensive groundfish conservation measures have been implemented, the latest of these occurring under Amendment 13 to the Multispecies Fisheries Management Plan (NEFMC 2009). A significant reduction in effort in the fishery is expected because of the Amendment 13 measures.

(2) Monkfish Fishery

The federal monkfish fishery occurs from Maine to the North Carolina/South Carolina border and is jointly managed by the New England Fishery Management Council (NEFMC) and Mid-Atlantic Fishery Management Council, under the Monkfish FMP (NEFSC 2005). Monkfish are primarily caught with bottom trawls and gillnets. Dredges also account for a small percentage of landings. The majority (73%) of all Atlantic sturgeon bycatch mortality in New England and Mid-Atlantic waters is attributed to the monkfish sink gillnet fishery (Shepherd et al. 2007). Observer data from 2001-2006 shows 224 recorded interactions between the monkfish fishery and Atlantic sturgeon, with 99 interactions resulting in death, a 44% mortality rate for Atlantic sturgeon that are taken as bycatch.

(3) Spiny Dogfish Fisheries

The primary gear types for the spiny dogfish fishery are sink gillnets, otter trawls, bottom longline, and driftnet gear (NEFSC 2003). Observer data from 2001-2006 shows 32 recorded interactions between the dogfish fishery and Atlantic sturgeon, with 5 interactions resulting in death; a 16% mortality rate for Atlantic sturgeon that are taken as bycatch (Shepherd et al. 2007).

(4) Atlantic Bluefish Fishery

The Atlantic bluefish fishery has been operating in the U.S. Atlantic for at least the last half century, although its popularity did not heighten until the late 1970s and early 1980s (MAFMC and ASMFC 1998). The gears used include otter trawls, gillnets, and hook-and-line. The majority of commercial fishing activity in the north Atlantic and Mid-Atlantic occurs in the late spring to early fall, when bluefish are most abundant in these areas (NEFSC 2005).

(5) Northeast Skate Fishery

The skate fishery has typically been composed of both a directed fishery and an incidental fishery. Otter trawls are the primary gear used to land skates in the U.S., with some landings also coming from sink gillnet, longline, and other gear (NEFSC 2007b). For Section 7 purposes, NMFS considered the effects to ESA-listed species of the directed skate fishery. Fishing effort that contributes to landings of skate for the incidental fishery is considered during Section 7 consultation on the directed fishery in which skate bycatch occurs.

(6) Atlantic Mackerel/Squid/Butterfish Fisheries

Atlantic mackerel/squid/butterfish fisheries are managed under a single FMP, which was first implemented on April 1, 1983. Trawl gear is the primary fishing gear for these fisheries, but several other types of gear may also be used, including hook-and-line, pot/trap, dredge, pound net, and bandit gear.

(7) Summer Flounder, Scup, and Black Sea Bass Fisheries

In the Mid-Atlantic, summer flounder, scup, and black sea bass are managed under a single FMP since these species co-occur and are often caught together. Otter trawl gear is used in the commercial fisheries for all 3 species. Floating traps and pots/traps are used in the scup and black sea bass fisheries, respectively (MAFMC 2007b).

4.4.1.2 Fisheries Independent Monitoring

NMFS Integrated Fisheries Independent Monitoring Activities in the Southeast Region promotes and funds projects conducted by the SEFSC and other NMFS partners to collect fisheries independent data. The various projects use a variety of gear (e.g., trawls, nets, etc.) to conduct fishery research. A Biological Opinion was issued for these activities in 2016 that concluded Atlantic sturgeon are likely to be adversely affected, but that these activities are not likely to jeopardize the species' continued existence. Atlantic sturgeon are incidentally taken during the course of these activities. Up to 4 Gulf of Maine DPS, 7 New York Bight DPS, 4 Chesapeake Bay DPS, 1 Carolina DPS, and 5 South Atlantic DPS Atlantic sturgeon lethal takes are expected over continuing 5 year periods.

In June 2016, NMFS completed a Programmatic Consultation on the Continued Prosecution of Fisheries and Ecosystem Research Conducted and Funded by the Northeast Fisheries Science Center that concluded Atlantic sturgeon are likely to be adversely affected, but that these activities are not likely to jeopardize the species' continued existence. Atlantic sturgeon are incidentally taken during the course of these activities. Up to 3 Gulf of Maine DPS, 15 New York Bight DPS, 4 Chesapeake Bay DPS, 1 Carolina DPS, and 7 South Atlantic DPS Atlantic sturgeon lethal takes are expected over continuing 5 year periods.

In January 2017, NMFS completed a consultation on USFWS funding the Georgia Department of Natural Resources Coastal Resources Division (GCRD) to collect, analyze and report biological and fisheries information to describe the conditions or health of recreationally important finfish populations and develop management recommendations that would maintain or restore the stocks in coastal Georgia. The Biological Opinion concluded that up to 2 Gulf of Maine DPS, 2 New York Bight DPS, 2 Chesapeake Bay DPS, 2 Carolina DPS, and 2 South Atlantic DPS Atlantic sturgeon lethal takes are expected over continuing 5 year periods, but that these adverse effects are not likely to jeopardize the continued existence of Atlantic sturgeon DPSs.

4.1.1.3 ESA Permits

Incidental Take Permit

The Opinion evaluating the incidental take permit for commercial shad fisheries in Georgia determined the operation of the fishery was likely to adversely affect Atlantic sturgeon but would not jeopardize its continued existence. NMFS determined that incidental capture by fisherman will be 140 Atlantic sturgeon per year in the Altamaha River, 35 Atlantic sturgeon per year in the Savannah River, and 5 Atlantic sturgeon per year in the Ogeechee River; the animals will be juveniles and subadults. The Biological Opinion anticipated the maximum intercept rate for each Atlantic sturgeon DPS to be: South Atlantic DPS 95%; Chesapeake Bay DPS 20%; Carolina DPS 15%; New York Bight DPS 10%; and Gulf of Maine DPS 2% of the total number of incidental capture, and a mortality rate of 1% (NMFS 2013c). Two years of data indicates that the number of incidental captures in Georgia's shad fisheries is less than anticipated. Subsequent to the completion of the Biological Opinion, the Ogeechee River was closed to commercial shad fishing in 2014.

ESA Section 10 Scientific Research Permit

Through issuance of ESA Section 10(a)(1)(A) permits, scientific and enhancement studies are conducted by researchers on Atlantic sturgeon. There are currently 3 Section 10(a)(1)(A) scientific research permits issued to study Atlantic sturgeon in the action area. The studies authorize researchers to anesthetize; collect eggs; attach external instrument (e.g., VHF, satellite); insert internal instrument, (e.g., VHF, sonic); mark, PIT tag; measure; photograph/video; fin clip; and weigh animals. Most takes authorized under these permits are expected to be nonlethal, but there are a few anticipated mortalities. Before any research permit is issued, the proposal must be reviewed under the permit regulations (i.e., must show a benefit to the species). In addition, since issuance of the permit is a federal activity, Section 7 analysis is also required to ensure the issuance of the permit is not likely to result in jeopardy to the species.

4.4.1.4 Dredging

Riverine, nearshore, and offshore areas are often dredged to support commercial shipping, recreational boating, construction of infrastructure, and marine mining. Dredging activities can pose significant impacts to sturgeon through direct capture. Environmental impacts of dredging that could also impact sturgeon include the following: (1) direct removal/burial of organisms; (2) turbidity/siltation effects; (3) contaminant resuspension; (4) noise/disturbance; (5) alterations to hydrodynamic regime and physical habitat; and (6) loss of riparian habitat (Chytalo 1996; Winger et al. 2000).

Maintenance dredging of federal navigation channels can adversely affect Atlantic sturgeon due to their benthic nature. Hydraulic dredges (e.g., hopper, cutterhead) can lethally harm sturgeon directly by entraining sturgeon in dredge drag arms and impeller pumps. Atlantic sturgeon mortalities in mechanical dredges (i.e., clamshell) have also been documented (Dickerson 2011). Potential impacts from hydraulic dredge operations may be avoided by imposing work restrictions during sensitive time periods (i.e., spawning,

migration, feeding) when sturgeon are most vulnerable to mortalities from dredging activity.

Dickerson (2011) summarized observed takings of 29 sturgeon from dredging activities conducted by the USACE off of the Atlantic coast and observed from 1990-2010: 2 Gulf, 11 shortnose, and 15 Atlantic, and 1 unidentified due to decomposition. Of the 3 types of dredges included (hopper, clamshell, and pipeline) in the report, most sturgeon were captured by hopper dredge. Notably, reports include only those trips when an observer was on board to document capture.

On November 4, 2011, NMFS completed an Opinion on the dredging and expansion of the Savannah Harbor (NMFS 2011a). The Opinion concluded that the project was not likely to jeopardize any ESA-listed species (including Atlantic sturgeon) if it implemented and complied with these mitigating measures:

- 1) Finalization of the off-channel rock-ramp fish passage design in coordination with NMFS and the other federal and state resource agencies.
- 2) Construction of the fish passage facility at the New Savannah Bluff Lock and Dam to provide access to historical spawning habitat for sturgeon as a mitigation measure.
- 3) Completion of the development and implementation of a comprehensive monitoring and adaptive management plan in coordination with NMFS and the other federal and state resource agencies to help insure the success of all mitigating measures including the fish passage facility.

The Opinion concluded that 4 Atlantic sturgeon would be killed as a result of interactions with dredges and another 20 would be taken in relocation trawlers but released alive. On October 13, 2017, NMFS completed a second amendment to the original Biological Opinion (SER-2010-05579) and revised their ITS for Atlantic sturgeon. The new ITS superseded the previous 2011 and 2013 ITSs Atlantic sturgeon and was issued for the entire project. NMFS determined the project is likely to adversely affect, but is not likely to jeopardize Atlantic sturgeon. The Opinion concluded that 17 Atlantic sturgeon would be killed as a result of interactions with dredges and another 198 would be taken in relocation trawlers (195 non-lethal, 3 lethal).

4.4.2 State Actions

4.4.2.1 Scientific Research

State Fisheries

Atlantic sturgeon are known to be adversely affected by gillnets and otter trawls. Given these gear types are most frequently used in state waters, state fisheries may have a greater impact on Atlantic sturgeon than federal fisheries using these same gear types.

Descriptions of Atlantic sturgeon captured in the South Atlantic shrimp fisheries operating in both federal and state waters is described previously in Section 4.4.1.1.

The commercial shad fisheries in Georgia incidentally capture Atlantic sturgeon. Georgia implemented regulations restricting fishing to the lower portions of the Savannah, Ogeechee, and Altamaha Rivers and close the fishery in the Satilla and St. Marys River to reduce sturgeon bycatch. The Georgia shad fishery is open from January 1 to as late as April 30 each year, but would typically end March 31. Georgia applied for, and received, an Incidental Take Permit from NMFS in 2013.

4.4.3 Marine Debris and Acoustic Impacts

A number of activities that may indirectly affect listed species in the action area of this consultation include anthropogenic marine debris and acoustic impacts. The impacts from these activities are difficult to measure or even to attribute to federal, state, local, or private actions. Where possible, conservation actions are being implemented to monitor or study impacts from these sources.

4.4.4 Marine Pollution and Environmental Contamination

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local or private action, may indirectly affect Atlantic sturgeon in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as: PCBs; storm water runoff from coastal towns, cities, and villages; and runoff into rivers that empty into bays and groundwater.

Atlantic sturgeon may be particularly susceptible to impacts from environmental contamination due to their benthic foraging behavior and long-life span. Sturgeon using estuarine habitats near urbanized areas may be exposed to numerous suites of contaminants within the substrate. Contaminants, including toxic metals, polychlorinated aromatic hydrocarbons (PAHs), organophosphate and organochlorine pesticides, polychlorinated biphenyls (PCBs), and other chlorinated hydrocarbon compounds can have substantial deleterious effects on aquatic life. Effects from these elements and compounds on fish include production of acute lesions, growth retardation, and reproductive impairment (Cooper 1989; Sindermann 1994).

Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but their long-term effects are not known (Ruelle and Henry 1992; Ruelle and Keenlyne 1993). Elevated levels of contaminants, including chlorinated hydrocarbons, in several other fish species are associated with reproductive impairment (Cameron et al. 1992; Drevnick and Sandheinrich 2003; Hammerschmidt et al. 2002; Longwell et al. 1992), reduced egg viability (Billsson et al. 1998; Giesy et al. 1986; Mac and Edsall 1991; Matta et al. 1997; Von Westernhagen et al. 1981b), reduced survival of larval fish (Berlin et al. 1981; Giesy et al. 1986), delayed maturity (Jorgensen et al. 2004a), and posterior malformations (Billsson et al. 1998). Pesticide exposure in fish may affect antipredator and homing behavior, reproductive function, physiological development, and swimming speed and distance (Beauvais et al. 2000; Moore and Waring 2001; Scholz et al. 2000; Waring and Moore 2004). Moser and Ross (1995) suggested that certain deformities and ulcerations found in Atlantic sturgeon in North Carolina's Brunswick River might be due to poor water quality in addition to possible boat-propeller-inflicted injuries. It should be noted that the

effect of multiple contaminants or mixtures of compounds at sublethal levels on fish has not been adequately studied. Atlantic sturgeon use marine, estuarine, and freshwater habitats and are in direct contact through water, diet, or dermal exposure with multiple contaminants throughout their range.

Sensitivity to environmental contaminants varies among fish species and life stages. Early life stages of fish seem to be more susceptible to environmental and pollutant stress than older life stages (Rosenthal and Alderdice 1976). In aquatic toxicity tests (Dwyer et al. 2000), Atlantic sturgeon fry were more sensitive to 5 contaminants (carbaryl, copper sulfate, 4-nonylphenol, pentachlorophenol, and permethrin) than fathead minnow (*Pimephales promelas*), sheepshead minnow (*Cyprinodon variegatus*), and rainbow trout (*Oncorhynchus mykiss*) - 3 common toxicity test species - and 12 other species of threatened and endangered fishes. The authors note, however, that Atlantic sturgeon were difficult to test and conclusions regarding chemical sensitivity should be interpreted with caution.

Another suite of contaminants occurring in fish are metals (mercury, cadmium, selenium, lead, etc.), also referred to as trace metals, trace elements, or inorganic contaminants. Post (1987) states that toxic metals may cause death or sublethal effects to fish in a variety of ways and that chronic toxicity of some metals may lead to the loss of reproductive capabilities, body malformation, inability to avoid predation, and susceptibility to infectious organisms.

Dioxin and furans were detected in ovarian tissue from shortnose sturgeon caught in the Sampit River/Winyah Bay system (S.C.). Results showed that 4 out of 7 fish tissues analyzed contained tetrachlorodibenzo-p-dioxin (TCDD) concentrations greater than 50 pg/g (parts-per-trillion), a level which can adversely affect the development of sturgeon fry (J. Iliff, NOAA, Damage Assessment Center, Silver Spring, M.D., unpublished data).

The EPA published its second edition of the National Coastal Condition Report (NCCR II) in 2004, which is a “report card” summarizing the status of coastal environments along the coast of the U.S. (EPA 2005). The report analyzes water quality, sediment, coastal habitat, benthos, and fish contaminant indices to determine status. The Southeast region (North Carolina - Florida) received an overall grade of B. There was a mixture of poor benthic scores scattered along the Southeast region.

4.4.5 Climate Change

As discussed earlier in this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. The effects of changes in water quality (temperature, salinity, DO, contaminants, etc.) in rivers and coastal waters inhabited by Atlantic sturgeon are expected to be more severe for those populations that occur at the southern extreme of the Atlantic sturgeon’s range, and in areas that are already subject to poor water quality as a result of eutrophication. As discussed in Section 3 of this Opinion, the South Atlantic and Carolina DPSs are within a region that will likely experience overall climatic drying. Atlantic sturgeon from these DPSs are already susceptible to reduced water quality resulting from various factors: inputs of nutrients;

contaminants from industrial activities and non-point sources; and interbasin transfers of water. Still, more information is needed to better determine the full and entire suite of impacts of climate change on Atlantic sturgeon and specific predictions regarding impacts in the action area are not currently possible.

4.4.6 Conservation and Recovery Actions Benefitting Atlantic Sturgeon

State and Federal Moratoria on Directed Capture of Atlantic Sturgeon

In 1998, the ASMFC instituted a coast-wide moratorium on the harvest of Atlantic sturgeon, which is to remain in effect until there are at least 20 protected age classes in each spawning stock (anticipated to take up to 40 or more years). NMFS followed the ASMFC moratorium with a similar moratorium on the harvest of Atlantic sturgeon in federal waters. Amendment 1 to ASMFC's Atlantic sturgeon FMP also includes measures for preservation of existing habitat, habitat restoration and improvement, monitoring of bycatch and stock recovery, and breeding/stocking protocols.

Use of TEDs in Trawl Fisheries

Atlantic sturgeon benefit from the use of devices designed to exclude other species from trawl nets, such as TEDs. TEDs and bycatch reduction device requirements may reduce Atlantic sturgeon bycatch in Southeast trawl fisheries (ASSRT 2007). NMFS has required the use of TEDs in southeast U.S. shrimp trawls since 1989 and in summer flounder trawls in the Mid-Atlantic area (south of Cape Charles, Virginia) since 1992 to reduce the potential for incidental mortality of sea turtles in commercial trawl fisheries. These regulations have been refined over the years to ensure that TED effectiveness is maximized through more widespread use, and proper placement, installation, floatation, and configuration (e.g., width of bar spacing). NMFS has also been working to develop a TED, which can be effectively used in a type of trawl known as a flynet, which is sometimes used in the Mid-Atlantic and Northeast fisheries to target sciaenids and bluefish. A top-opening flynet TED was certified in the summer of 2007, but experiments are still ongoing to certify a bottom-opening TED. All of these changes may lead to greater conservation benefits for Atlantic sturgeon.

4.5 Factors Affecting the Central and Southwest Atlantic DPS of Scalloped Hammerhead Shark within the Action Area

The following analysis examines actions that may affect the Central and Southwest Atlantic DPS of scalloped hammerhead shark and its environment specifically within the action area. The activities that shape the environmental baseline in the action area of this consultation are primarily federal and territorial fisheries. The Central and Southwest Atlantic DPS of scalloped hammerhead shark was recently listed and effects from federal fisheries are being evaluated through ESA Section 7 consultation as appropriate. The 2014 status review (Miller et al. 2014) serves the best source of information for threats to the species associated with federal fisheries.

4.5.1 Federal Actions

4.5.1.1 Federal Fisheries

Atlantic HMS- Pelagic Longline Fisheries for Swordfish and Tuna

Atlantic pelagic fisheries for swordfish and tuna are known to incidentally capture the Central and Southwest Atlantic DPS of scalloped hammerhead sharks (Miller et al. 2014). An analysis of observer logbooks for this fishery between 2005 and 2009 indicates approximately 181 hammerhead sharks (all species, not just scalloped hammerheads) were caught per year in the Atlantic (Miller et al. 2014). This value did not include dead discards for which scalloped hammerhead sharks were the second most discarded species in terms of weight (NMFS 2011). A separate consultation on the effect of HMS pelagic longline fishery on listed species, including the Central and Southwest Atlantic DPS of Atlantic sturgeon, is currently underway.

HMS-Atlantic Commercial and Recreational Fisheries for Shark, Swordfish, Tuna, and Billfish

HMS Atlantic commercial and recreational fisheries for shark, swordfish, tuna, and billfish, with the exception of the Atlantic pelagic longline fishery, are the subject of this consultation. Some of the federally-managed fisheries for Atlantic HMS occur in the Caribbean. As discussed in this Opinion, vertical line gears associated with certain fisheries (rod and reel, bandit, buoy gear, and handline) are likely to adversely affect the Central and Southwest Atlantic DPS of scalloped hammerhead shark, and may have affected the species in the past.

Caribbean Reef Fish Fisheries

The Central and Southwest Atlantic DPS of scalloped hammerhead sharks' susceptibility to capture in fishing gear indicates that Caribbean Reef Fish fisheries, managed under the Caribbean Reef Fish FMP, may affect the species. A consultation on the effect of these fisheries on listed species, including the Central and Southwest Atlantic DPS of Atlantic sturgeon, is currently underway.

4.5.1.2 Fisheries Independent Monitoring

NMFS Integrated Fisheries Independent Monitoring Activities in the Southeast Region promotes and funds projects conducted by the SEFSC and other NMFS partners to collect fisheries independent data. The various projects use a variety of gear (e.g., trawls, nets, etc.) to conduct fishery research. The 2016 Opinion concluded that the operation of FIM projects under the umbrella action is also not likely to jeopardize the continued existence of the Central and Southwest Atlantic DPS of scalloped hammerhead shark. Up to 1 lethal take is expected over continuing 5 year periods.

4.5.2 State or Private Actions

While the Final Listing Rule identified federal activities that may adversely affect Central and Southwest Atlantic DPS of scalloped hammerhead sharks, many of those activities, if conducted by state or private entities, are also likely to adversely affect the species.

Significant proportions of Puerto Rico and/or the USVI coasts have been degraded by inland hydrological projects, urbanization, agricultural activities, and other anthropogenic activities such as dredging, canal development, sea wall construction, and mangrove clearing. These activities have led to the loss and degradation of habitats potentially important to scalloped hammerhead sharks.

The capture of scalloped hammerhead sharks by anglers operating in the Commonwealth of Puerto Rico and Territorial Waters of the USVI is allowed. These activities may potentially impact the Central and Southwest Atlantic DPS of scalloped hammerheads.

4.5.3 Conservation and Recovery Actions Shaping the Environmental Baseline

Nationally, the U.S. has implemented significant laws specifically for the conservation and management of sharks: the Shark Finning Prohibition Act and the Shark Conservation Act. The Shark Finning Prohibition Act was enacted in December 2000 and implemented by final rule on February 11, 2002 (67 FR 6194), and prohibited any person under U.S. jurisdiction from: (i) Engaging in the finning of sharks; (ii) possessing shark fins aboard a fishing vessel without the corresponding carcass; and (iii) landing shark fins without the corresponding carcass. It also implemented a 5% fin to carcass ratio, creating a rebuttable presumption that fins landed from a fishing vessel or found on board a fishing vessel were taken, held, or landed in violation of the Act if the total weight of fins landed or found on board the vessel exceeded 5% of the total weight of carcasses landed or found on board the vessel. The Shark Conservation Act was signed into law on January 4, 2011, and implemented by final rule on June 29, 2016 (81 FR 42285), and, with a limited exception for smooth dogfish (*Mustelus canis*), prohibits any person from removing shark fins at sea, or possessing, transferring, or landing shark fins unless they are naturally attached to the corresponding carcass. As expected, U.S. exports of dried shark fins dropped significantly after the passage of the Shark Finning Prohibition Act. In 2011, with the passage of the U.S. Shark Conservation Act, exports of dried shark fins dropped again. Thus, although the international shark fin trade is likely a driving force behind the overutilization of many global shark species, including scalloped hammerhead sharks, the U.S. participation in this trade appears to be diminishing. In March 2013, at the CITES Conference of the Parties voted in support of listing three species of hammerhead sharks (scalloped, smooth, and great) in CITES Appendix II—an action that means increased protection, but still allows legal and sustainable trade. This CITES listing was effective as of September 14, 2014. Export of their fins requires permits that ensure the products were legally acquired and that the Scientific Authority of the State of export has advised that such export is not detrimental to the survival of the species. States have also enacted shark finning bans. The 2017 Shark Finning Report to Congress lists states that have enacted shark finning bans including Hawaii (2010), California (2011), Oregon (2011), Washington (2011), the Commonwealth of the Northern Mariana Islands (2011), Guam (2011), American Samoa (2012), Illinois (2012), Maryland (2013), Delaware (2013), New York (2013), Massachusetts (2014), Rhode Island (2016), and Texas (2016) (NMFS 2017).

4.6 Factors Affecting Oceanic Whitetip Sharks within the Action Area

The following analysis examines actions that may affect this species and its environment specifically within the action area. The activities that shape the environmental baseline in the action area of this consultation are primarily federal fisheries. The best available information on this species can be found in the status review (Hill and Sadovy de Mitcheson 2013), the Proposed Listing Rule (81 FR 42268, December 29, 2016) and the Final Listing Rule (83 FR 4153, January 30, 2018).

The potential stabilization of oceanic whitetip shark populations since the 1990s in the Northwest Atlantic Ocean and Gulf of Mexico occurred concomitantly with the first Federal FMP for Sharks in the Northwest Atlantic Ocean and Gulf of Mexico, which first directly managed oceanic whitetip shark under the pelagic shark group, and included regulations on trip limits and quotas. Management of the pelagic shark group, including oceanic whitetip sharks, has evolved under the 2006 Consolidated HMS FMP and amendments. This indicates the potential efficacy of these management measures for reducing the threat of overutilization of the oceanic whitetip shark population in this region; therefore, under current management measures, including the implementation of ICCAT Recommendation 10-07 described below, the threat of overutilization is not likely as significant in the action area relative to other portions of the species' range.

4.6.1 Federal Actions

4.6.1.1 Federal Fisheries

Presently, there are no Opinions evaluating the effects of federal actions on the oceanic whitetip shark in the action area. Oceanic whitetip sharks are managed under the pelagic sharks group under the 2006 Consolidated HMS FMP and amendments. Current authorized gear types for oceanic whitetip sharks include: Bottom longline, gillnet, rod and reel, handline, or bandit gear. Oceanic whitetip sharks may not be retained when pelagic longline gear is onboard or on recreational (HMS Angling and Charter headboat permit holders) vessels that possess tuna, swordfish, or billfish. Circle hooks are required in the recreational shark fishery and the directed commercial shark fishery. There is no commercial minimum size limit. The annual quota for pelagic sharks (other than blue sharks or porbeagle sharks) is currently 488 mt dressed weight. NMFS monitors landings within the different shark quota complexes throughout the year and will close the fishing season for a fishery when 80% of the respective quota has been landed or is projected to be landed and 100 percent of the quota is anticipated to be landed by the end of the year. Atlantic sharks and shark fins from federally permitted vessels may be sold only to federally permitted dealers. Logbook reporting is required for selected fishers with a federal commercial shark permit.

In the Northwest Atlantic, the oceanic whitetip has been caught commercially and incidentally as bycatch by a number of fisheries. Commercial landings of oceanic whitetip sharks in the U.S. Atlantic have been variable, but averaged approximately 1,077 lb (488.7 kg; 0.4887 mt) per year from 2003–2013. Although oceanic whitetip sharks have been prohibited on U.S. Atlantic commercial fishing vessels with pelagic longline gear onboard

since 2011, they are still caught as bycatch within the pelagic longline fishery, and also are caught with other gears and are occasionally landed. An Opinion on the HMS pelagic longline fisheries is currently underway. However, until that Opinion is complete, insufficient data and information exist to reliably specify how many animals are taken in various federal fisheries (beyond the fisheries analyzed in this Opinion; please see Section 5 for our effects analysis).

4.6.2 State or Private Actions

4.6.2.1 Fisheries

Anglers operating in the Commonwealth of Puerto Rico and Territorial Waters of the USVI are allowed to retain oceanic whitetip sharks while not in possession of tunas, billfish or swordfish.

4.6.3 Conservation and Recovery Actions Shaping the Environmental Baseline

In 2011, NMFS published final regulations to implement ICCAT Recommendation 10–07, which addressed oceanic whitetip sharks caught in association with ICCAT fisheries. That recommendation, and domestic implementing regulations, prohibit retention of oceanic whitetip sharks in the pelagic longline fishery and on recreational (HMS Angling and Charter headboat permit holders) vessels that possess tuna, swordfish, or billfish (76 FR 53652; August 29, 2011). The implementation of regulations to comply with ICCAT Recommendation 10–07 for the conservation of oceanic whitetip sharks is likely the most influential regulatory mechanism in terms of reducing mortality of oceanic whitetip sharks in the U.S. Atlantic. It should be noted that oceanic whitetip sharks are still occasionally caught as bycatch and landed in this region despite its prohibited status in ICCAT associated fisheries (NMFS 2012; 2014), as retention is permitted in other authorized gears other than pelagic longlines (e.g., gillnets, bottom longlines); however, these numbers have decreased. Prior to the implementation of the retention prohibition on oceanic whitetip, an analysis of the 2005–2009 HMS pelagic longline logbook data indicated that, on average, a total of 50 oceanic whitetip sharks were kept per year, with an additional 147 oceanic whitetip sharks caught per year and subsequently discarded (133 released alive and 14 discarded dead). Thus, without the prohibition, approximately 197 oceanic whitetip sharks could be caught and 64 oceanic whitetip sharks (32%) could die from being discarded dead or retained each year (NMFS 2011b). Since the prohibition was implemented in 2011, estimated commercial landings of oceanic whitetip declined from only 1.1 mt in 2011 to only 0.03 mt (dressed weight) in 2013 (NMFS 2012a; NMFS 2014a). From 2013–2014, NMFS reported a total of 81 oceanic whitetip pelagic longline interactions, with 83% (67 individuals) released alive and 17% (14 individuals) discarded dead (NMFS 2014; 2015).

While the retention ban for oceanic whitetip does not prevent incidental catch or subsequent at-vessel and post-release mortality, it likely provides minor ecological benefits to oceanic whitetip sharks via a reduction in overall fishing mortality in the Atlantic pelagic longline fishery (NMFS 2011b). As discussed in section 4.6.3, in addition to general commercial and recreational fishing regulations for management of HMS, the U.S.

has implemented significant national laws for the conservation and management of sharks: the Shark Finning Prohibition Act and the Shark Conservation Act. Thus, although the international shark fin trade is likely a driving force behind the overutilization of many global shark species, including the oceanic whitetip, the U.S. participation in this trade appears to be diminishing.

Overall, regulations to control for overutilization of oceanic whitetip sharks in U.S. waters, including fisheries management plans with quotas and trip limits, species-specific retention prohibitions in pelagic longline gear, and finning regulations are not in and of themselves inadequate such that they are contributing to the global extinction risk of the species. In fact, it is likely that the stable CPUE trend observed for the oceanic whitetip shark in the Northwest Atlantic is largely a result of the implementation of management measures for pelagic sharks under the 2006 Consolidated HMS FMP. However, because oceanic whitetip sharks are highly migratory and frequently move beyond the action area under U.S. jurisdiction, these regulatory mechanisms are limited on the global stage in that they only provide protections to oceanic whitetip sharks while in U.S. waters.

4.7 Factors Affecting Giant Manta Rays within the Action Area

The following analysis examines actions that may affect this species and its environment specifically within the action area. The activities that shape the environmental baseline in the action area of this consultation are primarily fisheries. The best available information on this species can be found in the status review (Miller and Klimovich 2017), the Proposed Listing Rule (82 FR 3694, January 12, 2017), and the Final Listing Rule (83 FR 2916, January 22, 2018).

Miller and Klimovich (2017) concluded that giant manta rays are at risk throughout a significant portion of their range, due in large part to the observed declines in the Indo-Pacific. Atlantic populations are likely small and sparsely distributed. Take and trade in U.S. waters were not identified as significant threats contributing to the species' status and the agency's listing determination. In areas where the species is not subject to fishing, population abundance may be stable (Miller and Klimovich 2017).

4.7.1 Federal Actions

4.7.1.1 Federal Fisheries

Presently, there are no Opinions evaluating the effects of federal fishery actions on the giant manta ray in the action area. Giant manta rays are not a federally managed species under any FMP.²³ Giant manta ray bycatch has been reported in the coastal migratory pelagic resources gillnet fishery, which operates in federal waters of the Gulf of Mexico and South Atlantic, the Gulf of Mexico reef fish bottom longline fishery, and the shark gillnet and shark bottom longline fisheries, thus we know these fisheries have had some effect on the species. The shark gillnet and shark bottom longline fisheries are subject to

²³ The Caribbean Fishery Management Council recently approved an FMP to manage certain resources within the U.S. EEZ surrounding Puerto Rico, and that FMP would manage giant manta rays, prohibiting harvest of the species. The FMP has not yet been approved by the Secretary of Commerce and has not yet been implemented.

this consultation, with their past effects part of this environmental baseline. Effects from the other fisheries are currently being evaluated through ESA Section 7 consultation. Prior to any complete ESA Section 7 consultation, insufficient data and information exist to specify how many animals are taken in various federal fisheries. At this time, the giant manta ray status report and final listing rule represent the best available information on the status of the species generally and within the action area. As stated in the status review and final listing rule, giant manta rays are sometimes caught as bycatch in the U.S. bottom longline and gillnet fisheries operating in the western Atlantic. However, given the low estimates of bycatch in U.S. fisheries, impacts from this mortality on the species are likely to be minimal.

4.7.2 State or Private Actions

While the Final Listing Rule identified federal activities that may adversely affect giant manta rays, many of those activities, if conducted by state or private entities, are also likely to adversely affect the species.

Significant proportions of the southeastern continental U.S., Puerto Rico, and/or the USVI coasts have been degraded by inland hydrological projects, urbanization, agricultural activities, and other anthropogenic activities such as dredging, canal development, sea wall construction, and mangrove clearing. These activities have led to the loss and degradation of habitats potentially important to giant manta rays.

4.7.2.1 Fisheries

Anglers fishing in non-federal fisheries are allowed to retain giant manta rays but it is unclear from survey data which species of ray has been caught as often unspecified rays are recorded.

4.7.3 Conservation and Recovery Actions Shaping the Environmental Baseline

Manta rays were included on Appendix II of CITES at the 16 Conference of the CITES Parties in March 2013, with the listing going into effect on September 14, 2014. Export of manta rays and manta ray products, such as gill plates, require CITES permits that ensure the products were legally acquired and that the Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species (after taking into account factors such as its population status and trends, distribution, harvest, and other biological and ecological elements). Although this CITES protection was not considered to be an action that decreased the current listing status of the threatened giant manta ray (due to its uncertain effects at reducing the threats of foreign domestic overutilization and inadequate regulations, and unknown post-release mortality rates from bycatch in industrial fisheries), it may help address the threat of foreign overutilization for the gill plate trade by ensuring that international trade of this threatened species is sustainable. Regardless, because the United States does not have a significant (or potentially any) presence in the international gill plate trade, we have concluded that any restrictions on U.S. trade of the giant manta ray that are in addition to the CITES requirements are not necessary and advisable for the conservation of the species.

5.0 Effects of the Action

In this section of our Opinion, we assess the effects of proposed operation of HMS fisheries, excluding pelagic longline fishery, on listed species that are likely to be adversely affected by this proposed action. The HMS gear types that are not likely to adversely affect these species are discussed in Section 3.2. The analysis in this section forms the foundation for our jeopardy analysis in Section 7.0. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Data are limited, so in some instances, we make assumptions to overcome the limits in our knowledge. Sometimes, the best available information may include a range of values for a particular aspect under consideration, or different analytical approaches may be applied to the same data set. In those cases, the uncertainty is resolved in favor of the species (House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)). NMFS generally selects the value that would lead to conclusions of higher, rather than lower, risk to endangered or threatened species. This approach provides the “benefit of the doubt” to threatened and endangered species.

We have not identified any effects that are caused by or result from the proposed action that would occur later in time. Such potential effects include aspects such as habitat degradation and reduction of prey/foraging bases. The operation of the HMS commercial and recreational fisheries analyzed in this Opinion (i.e., vessel operations, gear deployment and retrieval as described in Section 2.0) is not expected to impact the water column or benthic habitat in any appreciable way. Unlike mobile trawls and dredges that physically disturb habitat as they are dragged along the bottom, the gears used in the HMS fisheries are suspended in the water column and do not affect water column or benthic habitat characteristics. The fisheries’ target and bycatch species are not foraged on or a primary prey species for sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, or giant manta rays. Prey competition is not expected to be a factor for any of the protected species discussed in this Opinion.

Approach to Assessment

We began our analysis of the effects of the action by first reviewing what activities (e.g., gear types and techniques) associated with the proposed action are likely to adversely affect sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, or giant manta rays in the action area (i.e., what the proposed action stressors are). We next reviewed the range of responses to an individual’s exposure to that stressor, and the factors affecting the likelihood, frequency, and severity of exposure. Afterwards, our focus shifted to evaluating and quantifying exposure. We estimated the number of individuals of each species likely to be exposed and the likely fate of those animals.

Effects of the operation of the HMS commercial and recreational fisheries analyzed in this Opinion on threatened and endangered species stem primarily from interactions with fishing gear, which results in the catch, injury, and/or death of an individual listed species. Our analysis, therefore, assumed listed species are not likely to be adversely affected by a

gear type unless they come in physical contact with fishing gear. We also assumed the potential effects of each gear type are proportional to the number of interactions between the gear and each species. The following types of fishing gear authorized for use in the HMS commercial and recreational fisheries are analyzed in this Opinion: hook-and-line gear, gillnet, purse seine, speargun, and harpoon. In grouping these gears for the purposes of analyzing their effects on listed species, we further divided hook-and-line gear into vertical line gear (i.e., which we defined as green-stick gear, rod and reel, bandit, buoy gear, and handline) and bottom longline gear. Gillnet gear can be further divided into strike, sink gillnet, and drift gillnet. Section 2.0 describes these fishing gears and how commercial and recreational fishers may use them to target HMS species. Section 3.2 describes the HMS gear types that are not likely to adversely affect these species.

The other potential route of direct effects of the proposed action on listed species is via vessel interactions resulting in injury, and/or death of an individual. Fishing vessels actively fishing either operate at relatively slow speeds, drift, or remain idle, when setting, soaking, and hauling gear. Thus, any listed species in the path of a fishing vessel would be more likely to have time to move away before being struck. However, fishing vessels transiting to and from port or between fishing areas can travel at greater speeds, particularly recreational vessels, and thus have more potential to strike a vulnerable species than during active fishing. NMFS' STSSN data indicate that vessel interactions are believed to be responsible for a large number of sea turtles stranding within the action area each year, so it seems reasonable that the HMS fisheries subject to this consultation may be responsible for some of these interactions.

For gear analysis purposes, we generally evaluated the HMS fisheries subject to this consultation by looking at hook-and-line (i.e., bottom longline and vertical line gear) and gillnet gear separately. The likelihood, frequency, and severity of gear interactions is different for different species groups (i.e., for sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays). Also the type of fishing gear, area fished, and the manner/technique in which the gear is used all affect the potential likelihood, frequency, and severity of listed species interactions. We therefore organized our Effects section first by species group and then by gear type and/or user group to the extent the effects were different and we had data to distinguish them. For sea turtles, we also included a vessel strike analysis.

In conducting this consultation, we searched all available databases for all listed species interactions in HMS gear. This section details the information on interactions that have been documented for sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. The data we reviewed included data from the SEFSC, the NEFSC, GARFO database, LPS, and MRIP. Because the shark fisheries underwent major management changes in 2008 to prevent overfishing of and rebuild overfished Atlantic sharks (73 FR 40658, July 15, 2008), when possible, we used data since 2008 to evaluate the likelihood of listed species interactions in the fishery. For the rest of the HMS fisheries analyzed in this Opinion, major changes in operation have not occurred since the 2001 Opinion, so we considered all data available since the last consultation. We believe the HMS take rates and effort levels analyzed in this Opinion

will remain generally the same in the future in comparison to the take rates and effort from which we derived our incidental take estimates. Therefore, our 3-year take number estimates are based on assuming a similar level of take and effort in the future. Section 2 of this Opinion and NMFS (2017) provide more detailed information on fishing effort. We used standard mathematical rounding in this analysis to the tenth or hundredth decimal place depending on the data available. Numbers were always rounded up for the final 3 year ITS for each species.

5.1 Effects on Sea Turtles

Of the gear types used in the HMS fisheries by commercial and/or recreational fishers, we believe hook-and-line gear (with the exceptions of green stick) and gillnet gear may affect, and are likely to adversely affect sea turtles. This section focuses on evaluating the effects of certain HMS hook-and line-gear (rod and reel, bandit, buoy, handline, and bottom longline), gillnet gear, and vessel interactions on sea turtles.

5.1.1 Types of Interactions and General Effects from Hook-and-Line Gear and Gillnet Gear

Hook-and-line and gillnet gear are known to adversely affect sea turtles via hooking, entanglement, trailing line, and/or forced submergence. Upon retrieval of the gear, bycaught sea turtles may be found and released alive or found dead because of forced submergence. Sea turtles released alive may later succumb to injuries sustained at the time of catch or from exacerbated trauma from ingested fishing hooks and/or entangling lines or lines otherwise still attached when they were released. Of the sea turtles hooked or entangled that do not die from their wounds, some may suffer impaired swimming or foraging abilities.

The following discussion summarizes in greater detail the available information on how individual sea turtles are likely to respond to interactions with all types of hook-and-line and gillnet fishing gear.

Entanglement

Sea turtles are particularly prone to entanglement because of their body configuration and behavior. Records of stranded or entangled sea turtles reveal that hook-and-line gear can wrap around the neck, flippers (particularly front flippers), or body of a sea turtle and severely restrict swimming or feeding. Entangling gear can interfere with a sea turtle's ability to swim or impair its feeding, breeding, or migration and prevent its surfacing if the line gets caught on an object below the surface, causing it to drown. If the sea turtle is entangled when young, the fishing line becomes tighter and more constricting as the sea turtle grows, cutting off blood flow and causing deep gashes, some severe enough to remove an appendage.

Entanglements are expected to be more common on vertical line because it is generally lighter, more flexible gear; however, sea turtles have been found entangled in branchlines (gangions), mainlines, and float lines of longline gear as well. Observer data from the shark bottom longline fishery indicate sea turtles entangled in longline are most often

entangled around the neck and fore flippers (NMFS unpublished data). Gillnets can also adversely affect sea turtles via entanglement.

Hooking

Sea turtles are also injured and sometimes killed by being hooked. Sea turtles are either hooked externally in the flippers, head, shoulders, armpits, or beak (i.e., foul-hooked) or internally inside the mouth or, when the animal has swallowed the bait, in the gastrointestinal tract (Balazs et al. 1995). Observer data from the pelagic and shark bottom longline fisheries indicates entanglement and foul-hooking are the primary forms of interaction between leatherback sea turtles and longline gear, whereas beak and internal hooking is much more prevalent in hardshell sea turtles, especially loggerheads (NMFS unpublished data). Internal hooking of leatherback sea turtles is much rarer. Almost all interactions with loggerheads result from taking the bait and hook; only a very small percentage of loggerheads are foul-hooked externally or entangled.

Hooks swallowed by sea turtles are of the greatest concern. Their throats are lined with strong cone-shaped papillae directed towards the stomach (White 1994). The presence of these papillae in combination with an S-shaped bend in the throat makes it difficult to see swallowed hooks when looking through a sea turtle's mouth. Because of the shape of a sea turtle's digestive tract, deeply swallowed hooks are also very difficult to remove without seriously injuring the turtle. A sea turtle's throat is attached firmly to underlying tissue; thus, if a sea turtle swallows a hook and tries to free itself or is hauled on board a vessel, the hook can pierce the sea turtle's throat or stomach and can pull organs from their connective tissue. These injuries can cause internal bleeding or infections, both of which can kill the sea turtle.

If a hook does not lodge into, or pierce, a sea turtle's digestive organs, it can pass through the sea turtle entirely (Aguilar et al. 1995; Balazs et al. 1995) with little damage (Work 2000). For example, a study of loggerheads deeply hooked by the Spanish Mediterranean pelagic longline fleet found ingested hooks could be expelled after 53-285 days (average 118 days) (Aguilar et al. 1995). If a hook passes through a sea turtle's digestive tract without getting lodged, the hook probably has not harmed the turtle.

Trailing Line

Trailing line (i.e., line left on a sea turtle after it has been caught and released), particularly line from a swallowed hook, poses a serious risk to sea turtles. Line trailing from an ingested hook is also likely to be ingested, which may irritate the lining of the digestive tract. The line may cause the intestine to twist upon itself until it twists closed, creating a blockage ("torsion"), or it may cause a part of the intestine to slide into another part of intestine like a telescopic rod ("intussusception"), also leading to blockage. In both cases, death is a likely outcome (Watson et al. 2005). It may also prevent or hamper foraging, eventually leading to death. Trailing line may also become snagged on a floating or fixed object, further entangling a turtle and potentially slicing its appendages and affecting its ability to swim, feed, avoid predators, or reproduce. Sea turtles have been found with trailing gear that has been snagged on the bottom, or has the potential to snag, thus anchoring them in place (Balazs 1985b). Long lengths of trailing gear are likely to

entangle the sea turtle, eventually leading to impaired movement, constriction wounds, and potentially death.

Forced Submergence

Generally, when sea turtles dive, their bodies create energy for their cells in a process that uses oxygen from their lungs. Sea turtles that are stressed from being forcibly submerged due to entanglement, eventually use up all their oxygen stores. When their oxygen stores are used up, they begin to create energy via a process that does not require oxygen (i.e., anaerobic glycolysis). This process can significantly increase the level of a certain type of lactic acid in a sea turtle's blood (Lutcavage and Lutz 1997); if the level gets too high, it can cause death.

Numerous factors affect the survival rate of forcibly submerged sea turtles. It is likely that the speed at which physiological changes occur and how long they last are related to the intensity of struggling and how long the animal is underwater (Lutcavage and Lutz 1997). The size, activity level, and condition of the sea turtle; the ambient water temperature; and if multiple forced submergences have recently occurred all affect how badly an animal may be injured by forced submergence. Disease factors and hormonal status may also influence survival during forced submergence. Larger sea turtles are capable of longer voluntary dives than small sea turtles, so young sea turtles may be more vulnerable to the stress from forced submergence. The normal process for creating cellular energy happens more quickly during the warmer months. Because this process takes place more quickly, oxygen stores are also used more quickly, and anaerobic glycolysis may begin sooner.

Subsequently, the negative effects from forced submergence may occur more quickly during warm months. With each forced submergence event, the level of lactic acid in the blood increases and can require a long (up to 20 hours) time to return to normal levels. Sea turtles are probably more susceptible to dying from high levels of lactic acid if they experience multiple forced submergence events in a short period of time. Recurring submergence does not allow sea turtles to reduce high levels of lactic acid (Lutcavage and Lutz 1997). Stabenau and Vietti (2003) illustrated that sea turtles that are given time to stabilize their pH level after being forcibly submerged have a higher survival rate. How quickly this happens depends on the overall health, age, size, etc., of the sea turtle, time of last breath, time of submergence, environmental conditions (e.g., sea surface temperature, wave action), and the nature of any sustained injuries at the time of submergence (NRC 1990).

Effects from forced submergence are most likely expected to result from gillnet gear. Effects from forced submergence are expected to sometimes result from bottom longline gear interactions. Although there may be some stress associated with catch on vertical line gear, forced submergence and its effects on sea turtles are generally not expected to occur from this gear because of short soak times and because sea turtles likely are able to swim and reach the surface to breathe despite having gear attached. Forced submergence is not expected to occur when fishing with vertical line unless entangling lines are caught on an object below the surface and result in the sea turtle's inability to reach the surface and breathe.

5.1.2 Factors Affecting the Likelihood of Exposure of Sea Turtles to Hook-and-Line Gear

A variety of factors may affect the likelihood and frequency of listed sea turtle species interacting with hook-and-line gear. The spatial and temporal overlap between fishing effort and sea turtle abundance and sea turtle behavior may be the most evident variable involved in anticipating interactions. Other fishing related-factors that may influence the likelihood and frequency of hooking, entanglement, and forced submergence effects include gear characteristics (e.g., hook sizes, bait) and fishing techniques employed (e.g., soak times). Each of these factors and its potential influence is discussed briefly below.

Spatial/Temporal Overlap of Fishing Effort and Sea Turtles

The likelihood and rate of sea turtle hookings and/or entanglements in HMS hook-and-line gear is at least in part a function of the spatial and temporal overlap of sea turtle species and fishing effort. The more abundant sea turtles are in a given area where and when fishing occurs, and the more fishing effort in that given area, the greater the probability a sea turtle will interact with gear. Environmental conditions may play a large part in both where sea turtles are located in the action area and whether a sea turtle interacts with hook-and-line gear.

Different life stages²⁴ of sea turtles are associated with different habitat types and water depths. For example, pelagic stage oceanic juvenile loggerheads are found offshore closely associated with *Sargassum* rafts. As loggerheads mature, they begin to live in coastal inshore and nearshore waters foraging over hard- and soft-bottom habitats of the continental shelf (Carr 1986, Witzell 2002). Therefore, gear set closer to these areas is more likely to encounter adult loggerheads. Gear set further offshore in deeper colder waters is more likely to catch leatherbacks and juvenile loggerheads.

Hook Type

The type of hook (size and shape) used may also impact the probability and severity of interactions with sea turtles. The point of a circle hook is turned toward the shank, while the point of a J-hook is not. The configuration of a circle hook reduces the likelihood of foul-hooking interactions because the point of the hook is less likely to accidentally become embedded in a sea turtle's appendage or shell. In some fisheries, circle hooks are wide enough to actually prevent hooking of some sea turtles if the sea turtle cannot get its mouth around the hook (Gilman et al. 2006). Circle hook configuration also reduces the severity of interactions with sea turtles because it has a tendency to hook in the animal's mouth instead of its pharynx, esophagus, or stomach (Prince et al. 2002; Skomal et al. 2002).

Soak Time/Number of Hooks

Hook-and-line gear interactions with sea turtles may be affected by both soak time and the number of hooks fished, independent of overall fishing effort. The longer the soak time, the greater the chances a foraging sea turtle may encounter the gear, and the longer a sea

²⁴ For loggerheads, hatchlings generally range from 4.5-15 cm SCL; "oceanic juveniles" range from 15-63 cm SCL; "oceanic/neritic juveniles" range from 41-82 cm SCL; and adults range from 82-100+ cm SCL (Conant et al. 2009).

turtle may be exposed to the entanglement or hooking threat, increasing the likelihood of such an event's occurrence. For example, Carlson et al. 2016 found that soak time of shark bottom longline gear was found to predict at-vessel hooking mortality, with the median time for a sea turtle mortality to occur being 14 to 15 hours. Likewise, as the number of hooks in the water in a given area increases, so may the likelihood of an incidental hooking event.

Bait Type and Sea Turtle Feeding Habits

Sea turtles, particularly loggerhead sea turtles, may be attracted to and bite baited hooks. 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. Kemp's ridley sea turtles also feed on these species. Thus, loggerhead and Kemp's ridley sea turtles may be the species attracted to gear baited with these prey items. Green, hawksbill, and leatherback turtles may still also be attracted to fishing bait and have been caught on fishing hooks, but their feeding habits make it less likely. Green sea turtles become herbivorous as they mature, feeding on algae and sea grasses, but also occasionally consume jellyfish and sponges. The hawksbill's diet is highly specialized and consists primarily of sponges (Meylan 1988). Leatherbacks feed primarily on cnidarians (medusae, siphonophores) and tunicates, so they are less likely to pursue bottom longline gear bait.

Bait characteristics (e.g., the type, size, and texture of the bait) may also influence the likelihood and frequency of certain sea turtle species becoming incidentally hooked. For example, in pelagic longline fisheries there has been considerable success in reducing leatherback sea turtles caught by modifying bait usage, particularly replacing squid baits with mackerel (Watson et al. 2005). There are laboratory studies on the effect different bait characteristics have on loggerhead sea turtles' feeding behavior and preferences (Kiyota et al. 2004; Stokes et al. 2006).

5.1.3 Estimating Sea Turtle Incidental Catch in Shark Bottom Longline Gear

Observations of the shark-directed bottom longline fishery in the Atlantic and Gulf of Mexico have been conducted since 1994. From 1994 through 2001, observer coverage was voluntary but beginning with the 2002 fishing season, observer coverage became mandatory. Observer coverage from 1994 through the 1st trimester of 2005 was coordinated by the Commercial Shark Fishery Observer Program (CSFOP), Florida Museum of Natural History, University of Florida, Gainesville, Florida. Starting with the 2nd trimester season of 2005, responsibility for the fishery observer program was transferred to NMFS, SEFSC, Panama City Laboratory (Hale et al. 2009).

In 2008, shark fisheries underwent major management changes to prevent overfishing of and rebuild overfished Atlantic via implementation of Amendment 2 (73 FR 40658, July 15, 2008). Changes implemented included (among others) reductions in commercial quotas and commercial retention limits, the establishment of additional time/area closures for bottom longline gear, establishment of a shark research fishery which allows NMFS to select a limited number of commercial shark vessels on an annual basis to collect catch data and life history data for future stock assessments, and changes to which species could be kept by commercial fishermen. Specifically, only commercial fishermen participating

in the shark research fishery are allowed to land sandbar sharks (*Carcharhinus plumbeus*). In addition, all vessels selected to participate in the research fishery are required to carry an observer on all of their all trips. Vessels not participating in the shark research fishery are also required to carry observers if selected; the target coverage rate for non-research shark vessels is 5-10% (Carlson et al. 2017). Because the management changes in 2008 had major impacts on shark fisheries, we used observer data since 2008 to estimate sea turtle takes on shark bottom longlines as we believe it is most representative of future sea turtle takes.

5.1.3.1 Observer Data Summary

Observer data from 2008-2016 in the Gulf of Mexico, South Atlantic, and Mid-Atlantic Regions, indicate 35 loggerheads were observed caught on bottom longline gear targeting sharks (Carlson et al. 2017 and NMFS unpublished data).²⁵ This includes 33 loggerheads that were observed caught on bottom longlines in the shark research fishery ²⁶ and two loggerheads that were observed caught in the shark non-research bottom longline fishery. Of the 35 loggerheads observed caught, 12 were dead and 23 were released alive. There was also one Kemp's ridley observed caught and released alive in 2016 in the non-research shark bottom longline (Table 5.4).

Of the 36 sea turtles observed caught (35 loggerheads and 1 Kemp's ridley) on shark bottom longline gear from 2008-2016, size information was available for 35 records (34 of the loggerheads and the single Kemp's ridley). All of the loggerheads records for which size information was available for indicated a slight majority of loggerheads were adults (19/34), and the Kemp's ridley was a juvenile (NMFS unpublished data).

Table 5.4 Sea Turtles Caught in Shark Bottom Longline (BLL) Gear by Region, 2008-2016: Gulf of Mexico (GOM), South Atlantic (SA), or Mid-Atlantic Bight (MAB)

Fishery	Year	Quarter	Species	Area	Condition
SHX BLL - Research Fishery	2008	4	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2009	2	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2009	2	Loggerhead	GOM	Dead
SHX BLL - Research Fishery	2010	2	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2010	4	Loggerhead	SA	Alive
SHX BLL - Non-Research Fishery	2010	1	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2010	3	Loggerhead	GOM	Alive
SHX BLL - Research Fishery	2011	1	Loggerhead	GOM	Alive
SHX BLL - Research Fishery	2011	2	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2011	2	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2011	3	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2012	1	Loggerhead	SA	Alive

²⁵ 2016 was the most recent data year available at the time this analysis was conducted for this Opinion. Carlson et. al 2017 evaluates information from 2008-2015, and this information was supplemented with NMFS unpublished data from 2016.

²⁶ Although the shark research fishery is not restricted to using bottom longline gear, all participants to date have fished exclusively with bottom longline gear and we expect future participants to do the same.

SHX BLL - Research Fishery	2012	4	Loggerhead	SA	Alive
SHX BLL – Non-Research Fishery	2013	3	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2013	4	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2013	4	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2014	2	Loggerhead	GOM	Dead
SHX BLL - Research Fishery	2014	2	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2014	4	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2014	4	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2014	4	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2014	4	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2014	4	Loggerhead	MAB	Dead
SHX BLL - Research Fishery	2015	1	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2015	1	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2015	2	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2015	3	Loggerhead	SA	Alive
SHX BLL – Non-Research Fishery	2016	1	Kemp's Ridley	GOM	Alive
SHX BLL - Research Fishery	2016	2	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2016	2	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2016	2	Loggerhead	SA	Dead
SHX BLL - Research Fishery	2016	2	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2016	3	Loggerhead	SA	Alive
SHX BLL - Research Fishery	2016	4	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2016	4	Loggerhead	MAB	Alive
SHX BLL - Research Fishery	2016	4	Loggerhead	MAB	Alive

Source: Carlson et al. 2017 and NMFS SEFSC unpublished data

5.1.3.2 Sea Turtle Catch in Shark Bottom Longline Gear

In our 2012 Opinion, we presented a quantitative evaluation of the effects of shark bottom longline gear on sea turtle species based on SEFSC-estimated loggerhead take levels from 2007-2010 in shark bottom longline gear (Carlson and Richards 2011). Per SERO's request, Carlson et al. (2017) conducted an update to that take analysis using 2008-2015 shark bottom longline observer data. The loggerhead take estimates in Carlson et al. (2017) are the best available estimates for the number of loggerhead sea turtle takes on shark bottom longline gear in the shark non-research fishery and in the shark research fishery.

Carlson et al. (2017) estimated that the total number of loggerheads caught in the shark bottom longline, non-research fishery from 2008-2015 was 38.4 (95% CI = 0-249.9). Carlson et al. (2017) used a simple ratio estimator to represent bycatch rates as "CPUE = number of protected species/effort (e.g., number of hooks per set)." An estimate of uncertainty in these estimates was derived from bootstrap resampling of the calculated CPUE data set. Total estimated bycatch was calculated using a 3-year average of CPUE for all areas combined multiplied by the total fishery effort data reported to the Logbook Program for all areas (Carlson et al. 2017). Carlson et al. (2017) and Carlson and Richards (2011) include more detailed discussions of the data sources used, calculation methods, constraints of those methods, and the assumptions under which those calculations were made.

Over the same time period (2008-2015), an additional 25 loggerhead sea turtles were observed caught on bottom longlines in the shark research fishery. Of these, 4 were caught in the Gulf of Mexico, 14 were caught in the South Atlantic, and 7 were caught in the Mid-Atlantic Bight (Carlson et al. 2017). Because all trips in the shark research fishery were observed, the loggerhead catch observed in that fishery does not require extrapolation, and we assume the 25 loggerhead takes observed on bottom longlines in the shark research fishery were the only takes that occurred during that time period.

In total, Carlson et. al. (2017) estimated 63.4 loggerhead sea turtles were caught in bottom longline gear across the shark research and non-research fisheries from 2008-2015 (Table 5.5).

Table 5.5 Loggerhead Catch, 2008-2015: Shark Bottom Longline Gear

Year	Total*
<i>Estimated Catch for Non-Research Fishery</i>	
2008-2015	38.4
<i>Observed Catch in Shark Research Fishery</i>	
Year	Number
2008	1
2009	2
2010	3
2011	4
2012	2
2013	2
2014	7
2015	4
Total Observed	25
Total Estimated and Observed (2008-2015)	63.4
Annual Average	7.9

*Carlson et al. 2017 also provides the 95% CIs and CVs for the estimated catch in the non-research shark fishery

Source: Carlson et al. 2017

Estimated Non-Loggerhead Sea Turtle Takes in Shark Bottom Longline Gear

Above we presented the number of loggerhead sea turtles taken in shark bottom longline gear from 2008-2015. During that time period, no green, leatherback, Kemp's ridley or hawksbill sea turtles were observed in Atlantic shark bottom longline gear in either the research and non-research fisheries. As shown in Table 5.4, however, there was one Kemp's ridley observed in the non-research fishery in the Gulf of Mexico in 2016. This indicates that a Kemp's ridley sea turtle was caught in the past and could be caught in the non-research fishery again in the future. Also, only 5-10% of non-research shark vessel bottom longline sets are observed so interactions with other species may go completely unobserved. Because we know all listed sea turtle species are vulnerable to catch on hooks, we will assume that all listed sea turtle species will be taken in the shark non-research fishery using bottom longline gear.

We also will assume that all listed sea turtle species will be taken in shark research fishery using bottom longline gear. Unlike the non-research fishery, because the research fishery has 100% observer coverage, we know it has only caught loggerheads to date. However,

the Kemps' ridley take observed in the non-research fishery indicates that a Kemp's ridley take could occur in the research fishery in the future. Both the non-research and the research fishery target sharks in the same general areas and both have the potential to interact with all listed sea turtle species, given the species' vulnerability to catch on hooks. In addition, the shark research fishery has less effort than the non-research fishery, making it unlikely that all potential interactions have been observed. For these reasons, we estimated takes of all species, even though takes were not observed, for the research fishery.

To estimate the number of non-loggerhead sea turtles that will be taken in bottom longline gear targeting sharks in the research and non-research fisheries, we queried the Sea Turtle Salvage and Stranding Network's (STSSN) on-line database for the number of sea turtle strandings of each species that occurred in the Gulf of Mexico and South Atlantic from 2014-2018. We used the STSSN data as a proxy for relative species abundance in the action area. We also assumed that the probability of catching any sea turtle species was equal through time and space and solely a function of their relative abundance in the action area. We used 2014-2018 data, the most recent stranding data available, as we think it is informative as to the relative species abundance in the area in the future. Derived STSSN species abundances were 49.4 percent loggerheads, 15.1 percent Kemp's ridleys, 32.9 percent greens, 1.3 percent leatherbacks, and 1.3 percent hawksbills.

The ratios from the STSSN dataset were used to calculate the number of non-loggerhead species expected to be taken by shark bottom longline gear. The total number of all sea turtles (N_{Total}) taken annually was calculated by dividing the Carlson et al. (2017) annual estimate of loggerheads taken in Atlantic shark bottom longline gear in the research and non-research fisheries from 2008-2015 (7.9) ($N_{\text{BLL Lo}}$) by the loggerhead species abundance (STSSN_{Lo}) (49.4 percent). The number of each Kemp's ridley ($N_{\text{BLL Kr}}$), green ($N_{\text{BLL Gr}}$), leatherback ($N_{\text{BLL Le}}$) and hawksbill ($N_{\text{BLL Hwk}}$) taken was estimated by multiplying the respective species abundance (i.e., STSSN_{Kr}, STSSN_{Gr}, STSSN_{Le}, or STSSN_{Hwk}) by the total number of sea turtles taken annually (N_{Total}).²⁷ Table 5.6 reports the results of those calculations.

Table 5.6 Estimated Annual Sea Turtle Catch in Shark Bottom Longline Gear

Species	Estimated Annual Take
Loggerhead	7.9 ^a
Kemp's ridley	2.4
Green	5.3
Leatherback	.21
Hawksbill	.21
Total	16

^a Estimated by Carlson et al. (2017)

²⁷ 1) $N_{\text{BLL Lo}} = \text{STSSN}_{\text{Lo}} \times N_{\text{Total}}$; 2) $N_{\text{BLL Lo}} \times \text{STSSN}_{\text{Lo}}^{-1} = N_{\text{Total}}$; 3a) $N_{\text{BLL Kr}} = \text{STSSN}_{\text{Kr}} \times N_{\text{Total}}$; 3b) $N_{\text{BLL Gr}} = \text{STSSN}_{\text{Gr}} \times N_{\text{Total}}$; 3c) $N_{\text{BLL Le}} = \text{STSSN}_{\text{Le}} \times N_{\text{Total}}$; 3d) $N_{\text{BLL Hwk}} = \text{STSSN}_{\text{Hwk}} \times N_{\text{Total}}$

5.1.3.3 Estimating Immediate and Post-Release Mortality of Sea Turtles in Shark Bottom Longline Gear

Immediate Mortalities in Shark Bottom Longline Gear

Identifying the number of individuals that may die as a result of interactions with shark bottom longline gear is necessary to better assess the impacts of the action on the species when we conduct our jeopardy analysis. SEFSC updated this estimated level of sea turtle take and mortalities (Table 5.4) in shark bottom longline gear for use in this Opinion, using observer data from 2008-2016. Although our take estimates were based on 2008-2015 we used the additional data year (i.e., 2016) available for considering post-release mortality because we assume post-release mortality has no correlation to year and we wanted to use all of the data available to us. The information provided by the SEFSC on the non-research shark bottom-longline fishery indicates that 63% (12 of 19) of sea turtles have an immediate mortality (i.e., are dead when gear is retrieved or die shortly after). The information from the shark research fishery indicates that 30% of bycaught loggerheads between 2008 and 2016 were dead upon release (10 of 33) and 70% (23 of 33) were released alive. Since 100% of shark research fishery trips are observed, we believe this rate of mortality accurately reflects the number of animals that suffered immediate mortalities after interactions within the shark research fishery from 2008-2016. Therefore, we estimate a 30% immediate mortality rate for sea turtles caught with shark bottom longline gear from the research fishery, not the 63% from the non-research shark bottom longline fishery, because the information from the shark bottom longline research fishery reflects a larger sample size (33 interactions, not 19) with 100% observer coverage. Thus, we believe this information is more reliable for representing overall mortality rate.

Applying that mortality rate to our sea turtle take estimates, we estimate that a total of 2.4 loggerheads, 0.7 Kemp's ridley, 1.6 greens, 0.1 leatherback, and 0.1 hawksbill sea turtles may suffer immediate mortality annually after interacting with shark bottom longline gear. Table 5.7 summarizes those calculations and estimates.

Post-Release Mortalities in Shark Bottom Longline Gear

Most, if not all, sea turtles released alive from bottom longline gear will have experienced a physiological injury from forced submergence and/or traumatic injury from hooking and entanglement, and many may still carry penetrating or entangling gear. Thus, in addition to the mortality observed at the time of release, some level of post-release mortality is expected.

In January 2004, NMFS developed draft criteria for estimating post-release mortality of sea turtles, based on the best available information on the subject, to set standard guidelines for assessing post-release mortality from pelagic longline interactions. In 2006, those criteria were revised and finalized (Ryder et al. 2006). Under the revised criteria, overall mortality ratios are dependent upon the type of interaction (i.e., hooking, entanglement, etc.); the location of hooking, if applicable (i.e., hooked externally, hooked in the mouth, etc.); the amount/type of gear remaining on the animal at the time of release (i.e., hook remaining, amount of line remaining, entangled or not); and species (i.e., hardshells versus leatherbacks). Therefore, the experience, ability, and willingness of the crew to remove the

gear, and the availability of gear-removal equipment, are very important factors influencing post-release mortality. During real world application of these criteria (e.g., (Epperly and Boggs 2004)), it became clear that not every hooking scenario encountered could be categorized using the criteria. Thus, in August 2011, the SEFSC updated the 2006 criteria by adding three additional hooking scenarios. Consequently, those updates modified the layout of the post-release mortality table appearing in Ryder et al. (2006); a revised table can be found in NMFS SEFSC (NMFS 2012d).²⁸

Because we saw no reason why the same factors affecting post-release mortality of sea turtles hooked on pelagic longlines (interaction type and amount of gear remaining) would not apply to bottom longlines, we used these criteria to estimate the likely level of post-release mortality in shark bottom longline fisheries. Using the post-release mortality criteria from NMFS SEFSC (2012d), the SEFSC assigned injury categories and release conditions to the sea turtles that had been released alive after being caught in shark bottom longline gear between 2005 and 2016 (NMFS unpublished data). Applying the appropriate post-release mortality percentages from NMFS SEFSC (2012) to those injury categories and release conditions, we determined the number of animals that likely died of their injuries following their release. We estimated that the average post-release mortality rate for sea turtles released from shark bottom-longline gear between 2005 and 2016 was 20% (NMFS unpublished data). Table 5.7 includes our estimates of the animals we believe survived their interaction with the gear, the animals that died immediately following the interaction, and those that were released alive but later died as a result of injury (i.e., post-release mortality).

Table 5.7 Estimated Annual Take, Immediate, and Post-Release Sea Turtle Mortalities in Shark Bottom Longline Gear

Species	Estimated Annual Take	Immediate Mortalities *	Released Alive**	Post-Release Mortalities ***	Total Annual Mortalities
Loggerhead	7.9	2.4	5.5	1.1	3.5
Kemp's Ridley	2.4	0.7	1.7	0.3	1
Green	5.3	1.6	3.7	0.7	2.3
Leatherback	0.21	0.1	0.1	0.02	0.12
Hawksbill	0.21	0.1	0.1	0.02	0.12
Total	16	4.9	11.1	2.14	7.04

* Immediate mortalities = 30 percent of the estimated annual take

** Released alive = Estimated annual take minus immediate mortalities

*** Post-release mortalities = 20 percent of released alive

²⁸ Available at: http://www.sefsc.noaa.gov/turtledocs/UPR_SEFSC_PHMortality_2012.pdf

5.1.3.4 Estimated 3-Year Sea Turtle Takes and Mortalities in Shark Bottom Longline Gear

In the previous section, we concluded that 7.9 loggerhead sea turtles, 2.4 Kemp's ridley sea turtles, 5.3 green sea turtles, 0.21 leatherback sea turtles and 0.21 hawksbill sea turtles would be taken annually. Thus, every 3 years we estimate that shark bottom longline gear will catch 23.7 loggerhead sea turtles, 7.2 Kemp's ridley sea turtles, 15.9 green sea turtles, 0.6 leatherback sea turtle and 0.6 hawksbill sea turtle. Of the sea turtles caught every 3 years, 10.5 (3.5*3) loggerheads, 3 (1*3) Kemp's ridleys, 6.9 (2.3*3) greens, 0.4 (0.12*3) leatherback, and 0.4 (0.12*3) hawksbill sea turtle takes are estimated to result in mortality.

In conducting this consultation, we noted that the current criteria used to estimate post-release mortality do not consider any decompression sickness (DCS) effects on sea turtles. This is because DCS has only been recently recognized as a new pathological condition that can compromise post-release survivorship of incidentally caught sea turtles. Garcia-Parraga et al. (2014) documented for the first time DCS, a previously undescribed condition, in sea turtles incidentally caught by trawl and gillnet fisheries of the Valencian Community region of Spain. Because the shark bottom longlines are fished entirely in water less than 40 m depth on average, we do not believe DCS is as much of a concern as a mortality factor from this component. However, in the absence of data, we believe that in rounding all of our mortality estimates in our 3 year estimated take in table 5.7, we have already inflated our mortality estimates and thus provided a sufficient buffer for any additional mortality risk associated with DCS.

Table 5.8 Estimated 3-Year Sea Turtle Take and Mortalities in Shark Bottom Longline Gear

Species	Estimated 3-Year Take	Mortalities	Non-Mortalities
Loggerhead	23.7	10.5	13.2
Kemp's Ridley	7.2	3	4.2
Green	15.9	6.9	9
Leatherback	0.6	0.4	0.2
Hawksbill	0.6	0.4	0.2
Total	48	21.2	26.8

5.1.4 Estimating Sea Turtle Incidental Catch in Vertical Line Gear

In the 2001 Opinion, NMFS stated the potential for take in HMS fisheries other than the pelagic longline (which is not part of this opinion), the shark bottom longline, and shark gillnet fisheries is low. NMFS estimated take of no more than 3 sea turtles, of any species, in combination, per calendar year in these other HMS fisheries (*i.e.*, tuna purse seine, harpoon/hand gear fisheries, hook-and-line, etc.). As described in Section 3.2, NMFS believes HMS purse seine gear, harpoon, speargun gear, and green-stick gear are not likely to adversely affect sea turtles. Therefore, NMFS believes the adverse effects to sea turtles from the proposed action other than the shark bottom longline gear and gillnet gear are limited to certain HMS vertical hook-and-line gear (*i.e.*, rod and reel, bandit rigs, buoy

gear, and handline; hereafter in this section, HMS vertical line means only these hook-and-line gears, which are likely to adversely affect sea turtles).

In conducting this consultation, we searched for new data on which to update our previous estimated sea turtle catch rates and the number of sea turtles caught that were attributed to commercial fishing with vertical lines to target HMS. We found no new records of sea turtle interactions on HMS commercial vertical lines. However, as we stated in the 2001 Opinion, we know sea turtles have at least been caught in other commercial rod-and-reel fisheries albeit at relatively low rates.

We also searched for new data on recreational vertical line sea turtle takes that could be used to estimate sea turtles takes in HMS recreational vertical line fisheries. In 2010, we initiated collaboration with other NMFS offices to improve the data available on which we can estimate and monitor sea turtle catch in recreational fisheries. In January 2010, the NMFS Office of Science and Technology (OST) agreed to lead a SERO and SEFSC team to develop possible survey designs and evaluate their effectiveness for gathering information on recreational fishing. To date, the team has developed and piloted two surveys related to recreational fishing: a supplemental mail survey for private vessels surveyed via MRIP, and a charter headboat survey. Both surveys were conducted in North Carolina. At this time, further analysis of these studies needs to be completed to better understand how to move forward to collect data that can be expanded to a wider universe than sampled.

In summary, there is no additional information on which to base HMS commercial vertical line takes of sea turtles since the 2001 Opinion. In addition, recreational sea turtle interaction surveys conducted since our 2001 Opinion are too limited in scope, and STSSN stranding data associated with vertical line are too broad to produce estimates of the number of sea turtle hookings or entanglements by recreational HMS fishing in federal waters. Therefore, we believe the take estimate used in the 2001 Opinion (3 sea turtles) in the hand gear and rod-and-reel fisheries²⁹ is the best available estimate for sea turtle take in vertical line gear as defined above (i.e., rod and reel, bandit, buoy gear, and handline). We use the take estimates from the 2001 Opinion for all species of sea turtles and then estimate take for the different sea turtle species based on the approach we used to estimate take of different sea turtle species for bottom longline gear (in the prior section) and gillnet gear (discussed below). Specifically, we assumed that the probability of catching any sea turtle species was equal through time and space and solely a function of their relative abundance in the action area. We relied on the sea turtle species relative abundance estimates calculated from the STSSN to estimate the number of loggerhead, green, leatherback,

²⁹ As defined in the 2001 Opinion, hand gear included hook and line (i.e., rod and reel, bandit), handline, and harpoon gears. Before 2007, buoy gear catch data was included as handline catch data. The take estimate noted above for “hand gear and rod and reel fisheries,” applied to harpoon, purse seine, rod and reel, bandit gear, and handline (which included buoy gear), however, NMFS only expected take from rod and reel, bandit, and handline (including buoy gears), and harpoon gears. In this Opinion, we concluded that harpoon gear is not likely to adversely affect sea turtles; the remaining gears (rod and reel, bandit, handline, and buoy gear) are the gears we expect to be likely to adversely affect sea turtles in this Opinion. Thus the take estimate for the hand gear and rod and reel fisheries in the 2001 Opinion is applicable to the take estimate from certain vertical line gears (rod and reel, bandit, buoy, and handline) in this Opinion.

Kemp's ridley, and hawksbill sea turtle takes in vertical line gear. Those numbers are presented below in Table 5.9. We queried the STSSN on-line database for the number of sea turtle strandings of each species that occurred in the Gulf of Mexico and South Atlantic from 2014-2018. We used 2014-2018 data, the most recent stranding data available, as we think it is informative as to the relative species abundance in the area in the future. The ratios from STSSN dataset were used to calculate the number of loggerhead and non-loggerhead species taken. Derived STSSN species abundance were 49.4% loggerheads, 32.9% green, 1.3% leatherbacks, 1.3% hawksbills, and 15.1% Kemp's ridleys. We applied these ratios to the 3 total turtles taken in vertical line gear.

Table 5.9 Estimated Annual Sea Turtle Incidental Catch in Vertical Line Gear

Species	Estimated Annual Take
Loggerhead	1.5
Kemp's ridley	0.45
Green	0.99
Leatherback	0.04
Hawksbill	0.04
Total	3

5.1.4.1 Estimating Immediate and Post-Release Mortality of Sea Turtles in Vertical Line Gear

As discussed in 5.2.1, sea turtle mortality can occur prior to release (i.e., immediate mortality) or later in time (i.e., post-release mortality). Below, both types of mortality are reviewed and estimated for vertical line interactions.

Immediate Mortality in Vertical Line Gear

We believe all sea turtles caught with HMS vertical line gear (as defined above, i.e., rod and reel, bandit, buoy, and handline), whether fished recreationally or commercially, are released alive because: (1) sea turtles can very likely breath-hold longer than typical soak times (less than one hour), even under stress, and (2) forcible submergence is extremely unlikely to occur as, except in cases of extreme entanglement (such as hooking late in a sea turtle's dive, combined with bottom-fouling or extremely heavy sinkers with very small sea turtles), hooked sea turtles will be able to surface and breathe. Based on that information, we believe it is highly unlikely that a sea turtle caught on a vertical line in the Atlantic HMS fisheries would be dead upon retrieval of the line, and we assume no immediate mortality.

Post-release Mortality in Vertical Line Gear

Post-release mortality criteria specific to sea turtles caught on vertical line interactions do not exist. We presume that sea turtles caught on vertical line gear and released alive would be in better overall health than if released alive from bottom longline gear because of the much shorter soak times and the animals' likely ability to reach the surface of the water to breathe. However, we see no reason why the same factors affecting post-release mortality of sea turtles hooked on bottom longlines (hook location and amount of gear remaining) would not apply to sea turtles hooked on vertical line gear. We assume sea turtles are, and

will continue to be, hooked in the jaw and released still hooked and with trailing line. We base this assumption on mainly circle hook use and anecdotal information that indicated fishers typically just cut the line when sea turtles are caught. In the absence of other quantitative data, we conservatively applied the same post-release mortality rates (i.e., 20%) to our commercial and recreational HMS vertical line take estimates as we applied to take in from shark bottom longline gear.

5.1.4.2 Estimated 3-Year Sea Turtle Takes and Mortalities in HMS Vertical Line Gear

In the previous section, we concluded that 1.5 loggerhead sea turtles, 0.45 Kemp's ridley sea turtles, 0.99 green sea turtles, 0.04 leatherback sea turtle, and 0.04 hawksbill sea turtles would be taken annually in vertical line gear. Therefore, every 3 years we expect 4.5 loggerhead sea turtles, 1.4 Kemp's ridley sea turtles, 3 green sea turtles, 0.1 leatherback sea turtles, and 0.1 hawksbill sea turtles will be caught in vertical line gear. Next, we applied the 20 percent post-release mortality rate to our 3-year catch estimates. Of the 4.5 loggerheads, 1.4 Kemp's ridleys, 3 greens, 0.1 leatherback, and 0.1 hawksbill sea turtles to be caught every 3-years, 0.9 (4.5×0.2) loggerheads, 0.3 (1.4×0.2) Kemp's ridleys, 0.6 (3×0.2) greens, 0.02 (0.1×0.2) leatherback, and 0.02 (0.1×0.2) hawksbill sea turtle takes are estimated to result in mortality (Table 5.10).

Table 5.10 Estimated 3-Year Sea Turtle Take and Immediate and Post-Release Mortalities in HMS Vertical Line

Species	Estimated 3-Year Take	Immediate Mortalities	Post-Release Mortalities	3-Year Total Mortalities
Loggerhead	4.5	0	0.9	0.9
Kemp's Ridley	1.4	0	0.3	0.3
Green	3	0	0.6	0.6
Leatherback	0.1	0	0.02	0.02
Hawksbill	0.1	0	0.02	0.02
Total	9.1	0	1.8	1.8

5.1.5 Factors Affecting the Likelihood of Exposure of Sea Turtles to Gillnet Gear

Entanglement/Forced Submergence

Gillnets can adversely affect sea turtles via entanglement and forced submergence. While the mechanism of catch is different between bottom longline and gillnet gears, many of the effects are the same. See Section 5.1.2 for the previous discussion on the effects of entanglement and forced submergence on sea turtles.

Net profile

Both length and profile (i.e., the percentage of the water column spanned by the net) of gillnets in the water column affect the likelihood of sea turtle exposure to gillnets. Gillnets spanning the entire water column (i.e., surface to bottom) are more likely to catch sea turtles than low-profile gillnets spanning only a narrow portion of the water column. For

example, drift gillnet gear is generally fished at the surface, while strike gillnet gear generally spans the entire water column to reduce fish loss from fish swimming under or over the net (Carlson and Bethea 2007).

Mesh Size

All mesh sizes are known to entangle sea turtles, but entanglement risks appear to increase with increasing mesh size. Smaller sea turtles may be more susceptible to entanglement in gillnets with smaller mesh sizes than are larger sea turtles.

Soak Times

The length of time gillnet gear is left in the water is another important consideration. The longer the soak time, the higher the likelihood sea turtles may encounter the gillnet gear and become entangled. Incidental catch of sea turtles is most frequently documented in long sets, and in lost or broken-off gear presumed to have been soaking for a long time.

Species Morphology

Sea turtles are prone to entanglement as a result of their body configuration and behavior. Records of stranded or entangled sea turtles reveal gear can wrap around the neck, flipper, or body of a sea turtle. These entanglements can severely restrict swimming or feeding.

Environmental Conditions

Environmental conditions may also play a large part in whether ESA-listed species interact with gillnet gear. Fishing gear can drift according to oceanographic conditions, including wind and waves, surface and subsurface currents, etc.; therefore, depending on these species' behavior, environmental conditions, and location of the set, ESA-listed species may become entangled in the gear.

Sea turtles also appear to associate with particular sea surface temperatures. From 1995-2006, observers aboard vessels fishing with gillnet gear in the Mid-Atlantic observed the incidental catch of 41 loggerhead, 5 green, 8 Kemp's ridley, and 5 leatherback sea turtles. The average sea surface temperature of loggerhead, green, and Kemp's ridley sea turtles observations was approximately 17°C; leatherbacks were found in cooler waters, averaging approximately 15°C ((Murray 2009a)). This distribution indicates fishing effort in cooler waters is more likely to take leatherback sea turtles and fishing in warmer waters increases the likelihood of interactions with loggerhead, green, and Kemp's ridley sea turtles.

5.1.6 Estimating Sea Turtle Incidental Catch in Southeast Shark Gillnet Gear

The shark fishery discussed in this section is the shark drift gillnet fishery that developed off the east coast of Florida and Georgia in the late 1980s and its history and observer requirements are described (Trent et al. 1997, Passerotti et al. 2011 and references therein, Carlson and Richards 2011). Since the implementation of Amendment 2 to the 2006 Consolidated Atlantic HMS FMP in 2008 (NMFS 2008a), the directed large coastal LCS gillnet fishery has been greatly reduced. The LCS trip limit implemented via Amendment 2 has essentially ended the strikenet fishery for LCS and limited the number of fishermen targeting LCS with drift gillnet gear. The SCS fishery was also limited by Amendment 2, but was more directly impacted by Amendment 3 to the 2006 Consolidated HMS FMP

(NMFS 2010a), which significantly reduced the SCS quota and established an individual quota for blacknose sharks. As a result, many gillnet fishermen that historically targeted sharks are now targeting non-HMS finfish species such as Spanish mackerel, king mackerel, and bluefish with varying types of gillnet gear (Passerotti et al. 2010).

5.1.6.1 Observer Data Summary

Table 5.11 summarizes the sea turtles observed caught from 2007 through 2010 by all shark gillnet gears used in the Southeast (sink, drift, and strike gillnets). Three loggerhead sea turtles were caught in sink gillnet gear in 2007 (Baremore et al. 2007). In 2009, one Kemp's ridley sea turtle was caught in drift gillnet gear (Carlson and Richards 2011). No sea turtle interactions were observed in 2008 (Passerotti and Carlson 2009), 2010 (Carlson and Richards 2011), or 2011 through 2015 (Carlson et al. 2017).

Table 5.11 Observed Sea Turtle Catch in Southeast Shark Gillnet Gear (2007-2010)

Fishery	Year	Species	Area	Condition
SHX – Sink	2007	Loggerhead	SA	Dead
SHX – Sink	2007	Loggerhead	SA	Alive
SHX – Sink	2007	Loggerhead	SA	Alive
SHX – Drift	2009	Kemp's ridley	GOM	Alive

Source: Carlson and Richards 2011

Size data is available for all three loggerheads observed caught by gillnet gear in 2007. Two of the three loggerheads were likely oceanic/neritic juveniles with sizes of 70.5 and 75.5 cm CCL; the third was likely an adult (86.8 cm CCL) (NMFS unpublished data).³⁰ The Kemp's ridley measured 19.4 cm CCL and was most likely a juvenile.³¹

5.1.6.2 Extrapolated Catch of Sea Turtles in Southeast Shark Gillnet Gear

In our analysis below, we rely on the calculated take numbers for the data from 2007-2010, discussed in Carlson and Richards 2011. Based on observer reports, we will not be using data from 2011-2015, discussed in Carlson et al. 2017, which reported no sea turtle interactions, as nearly all of the shark gillnet observer coverage was on strike gillnets, which we believe have a low chance of catching sea turtles, and thus are not informative of potential take in the sink and drift gillnet fisheries. In addition, as discussed, we believe future effort will be largely be with sink and drift gillnets. Though effort in the sink and drift gillnet fisheries for LCS and SCS has declined since 2008, there is still some effort and thus the potential for interactions.

³⁰ Loggerheads hatchlings generally range from 4.5-15 cm SCL; "oceanic juveniles" range from 15-63 cm SCL; "oceanic/neritic juveniles" range from 41-82 cm SCL; and adults range from 82-100+ cm SCL (Conant 2009).

³¹ Kemp's ridley sea turtles are considered adults at 60+ cm SCL (Ogren 1989).

Carlson and Richards (2011) estimated the total loggerhead and Kemp’s ridley sea turtles taken in the shark gillnet fisheries (regardless of type of gillnet or location within the Southeast) from 2007 through 2010 as 35.6 loggerheads and 11.8 Kemp’s ridleys, based on interactions with sink and drift gillnets.

Carlson and Richards (2011) employed a simple ratio estimator to represent bycatch rates (i.e., CPUE) of sea turtles in shark gillnet gear. More specifically, CPUE was calculated by dividing the sum of all observed sea turtles caught by species by the sum of the observed sets by gear type (i.e., sink, strike, or drift) (Snedecor and Cochran 1967). All observer data was combined (2007-2010) and stratified to the South Atlantic and Gulf of Mexico. Uncertainty in these estimates was derived from bootstrap re-sampling of the calculated CPUE data set (see Carlson and Richards 2011 for further discussion of the methods used to calculate uncertainty).

The incidental take estimates were calculated by multiplying the CPUE from the observer database by the total number of reported sets. Estimates were calculated for the Gulf of Mexico and South Atlantic regions and also for each gillnet gear type used in each region. An incidental take estimate for the entire shark gillnet fishery (i.e., both regions and all gear type combined) was also calculated by using the CPUE average for all areas combined multiplied by the total effort determined for all areas (Carlson and Richards 2011).

The total effort data used reflects all 2007-2010 gillnet trip reports received by the Coastal Fisheries Logbook Program (CFLP).³² The target species for each trip was determined by using the proportion of shark catch relative to the rest of the species landed for a trip. When sharks comprised 66.6% or more of a trip’s total catch it was considered a shark directed trip. When sharks accounted for less than 33.3% of the total catch it was considered “other”; shark landings between 33.3 and 66.6% of the total catch were considered “mixed.” Smooth dogfish were included with all other sharks for trip target determination (Carlson and Richards 2011). Carlson and Richards (2011) estimate that from 2007-2010, across all gillnet gear types (i.e., sink, drift, and strike), a total of 35.6 loggerhead interactions and 11.8 Kemp’s ridley interactions occurred. By gear type, 23.7 loggerhead interactions occurred in sink gillnets and 11.9 occurred in drift gillnets. All Kemp’s ridley catch occurred in drift gillnets (Carlson and Richards 2011). Table 5.12 summarizes these calculations.

Table 5.12 Estimated Sea Turtle Incidental Catch in Southeast Shark Gillnet Gear

Species	Gear Type	Estimated Take
Loggerhead	Sink Gillnet	23.7
	Drift Gillnet	11.9
	Total Estimated Takes (2007-2010)	35.6

³² In 2007, the CFLP began using an updated trip report form that provided gillnet fishermen a place to note the type of gillnet used (strike, drift, anchor, or other) as well as space to provide the number of sets. These fields were unavailable on logbook forms prior to 2007. There are some instances where fishermen have submitted a 2007 or later trip on a pre-2007 form (Carlson and Richards 2011).

	Average Annual Takes (2007-2010)	8.9
Kemp's Ridley	Drift Gillnet	11.8
	Total Estimated Takes (2007-2010)	11.8
	Average Annual Takes (2007-2010)	3

Source: Carlson and Richards 2011, NMFS unpublished data

We recognize that there have been changes to how the gillnet fisheries operate since the 2007-2010 time period on which the Carlson and Richards (2011) take estimates are based. For example, quotas implemented in Amendment 2 to the 2006 Consolidated HMS FMP reduced the use of strike gillnet gear for LCS. In addition, Amendments 2 and 3, in limiting the number of fishermen targeting LCS with all gillnet gear, reducing the quota for SCS, and establishing a quota for blacknose sharks, likely limited effort in the sink and drift gillnet fisheries, reducing the potential for interactions. Specifically, since 2012, the number of vessels that have used gillnet gear in the southeast and Gulf of Mexico and that have caught sharks, other than smooth dogfish, has ranged from the low twenties to thirty-six, and the number of trips per year has declined from over 400 in 2012 to just over 200 in 2018. Still, while effort is down, there is still effort in the sink and drift gillnet fisheries, and thus there is still the potential for take. Thus, we will rely on the estimates from Carlson and Richards 2011 as the best available scientific and commercial data available to estimate take in Southeast shark gillnet gear.

Estimated Non-Loggerhead and Non-Kemp's Ridley Incidental Catch in Southeast Shark Gillnet Gear

Since no green, leatherback, or hawksbill sea turtles were observed caught in Southeast shark gillnet gear from 2007-2010, there are no extrapolated take estimates in Carlson and Richards (2011). However, previous catch of these species in this gear type have been documented (see Carlson 2001, Carlson and Baremore 2002, Garrison 2007). Since we believe these sea turtles can be caught in shark gillnet gear, we also estimated takes of these species.

We used the same approach and assumptions discussed in Section 5.1.3.2, above, to calculate the number of green, leatherback, and hawksbill sea turtles caught in the Southeast shark gillnet gear in this section. The number of each green ($N_{GN\ Gr}$), leatherback ($N_{GN\ Le}$) and hawksbill ($N_{GN\ Hwk}$) taken was estimated by multiplying the respective species abundance (i.e., $STSSN_{Gr}$, $STSSN_{Le}$, or $STSSN_{Hwk}$) by the total number of sea turtles taken annually. We used the estimate of the number of loggerhead sea turtles taken annually from Carlson and Richards (2011) (8.9) and the SSTN abundance of loggerhead sea turtles to estimate the total number of sea turtles taken annually.³³ Derived STSSN species abundance were 49.4% loggerheads, 32.9% green, 1.3% leatherbacks, 1.3% hawksbills and 15.1% Kemp's ridleys. Since Carlson and Richards (2011) provided an

³³ 1) $N_{GN\ Lo} = STSSN_{Lo} \times N_{Total}$; 2) $N_{GN\ Lo} \times STSSN_{Lo}^{-1} = N_{Total}$; 3a) $N_{GN\ Gr} = STSSN_{Gr} \times N_{Total}$; 3b) $N_{GN\ Le} = STSSN_{Le} \times N_{Total}$; 3c) $N_{GN\ Hwk} = STSSN_{Hwk} \times N_{Total}$

estimate of loggerhead and Kemp's ridley takes we did not estimate them here. Table 5.13 reports the results of those calculations.

Table 5.13 Estimated Annual Sea Turtle Take in Southeast Shark Gillnet Gear

Species	Estimated Annual Take
Loggerhead	8.9 ^a
Kemp's ridley	3 ^a
Green	5.9
Leatherback	0.23
Hawksbill	0.23
Total	18.26

^a Estimated by Carlson and Richards 2011

5.1.6.3 Estimating Immediate and Post-Release Mortality of Sea Turtles in Southeast Shark Gillnet Gear

Carlson and Richards (2011) reported the final condition (i.e., alive or dead) of the observed bycatch events, but did not calculate mortalities. Identifying the number of individuals that may die as a result of interactions with shark gillnet gear is necessary to better assess the impacts of the action on the species when we conduct our jeopardy analysis. Observer information provided indicate that 33 turtles were caught in shark gillnet gear from 2001-2009. These reports indicate that 8 turtles, or 24% ($8/33 = 0.24$) of sea turtles caught, suffer an immediate mortality (i.e., are dead when gear is retrieved or die shortly after) (NMFS unpublished data).³⁴ Carlson and Richards (2011) indicate that during this time period, distinguishing gillnet type was difficult as 4-10% of records were reported as gillnet "other" which makes it difficult to distinguish category. In addition, fishers were still reporting (up to 44% of effort by year) to strikenet (reported as "drift, runaround") for sharks although the observer program indicated this activity had significantly decreased.

By definition, drift gillnets are not anchored and this configuration is likely more conducive to sea turtles being able to reach the surface to breathe. Because sink gillnets are weighted, entangled sea turtles may have a more difficult time reaching the surface to breathe. Thus, sea turtles entangled in drift gillnet gear may be more likely to survive an entanglement than one entangled in a sink gillnet. In theory, by applying this mortality rate to all gillnets, including drift gillnets, we maybe overestimating mortality. However, we know that sea turtle mortalities have occurred as a result of drift gillnet entanglements in other fisheries (see Carlson 2000, Carlson 2001, Carlson and Baremore 2002, Garrison 2003, Carlson and Bethea 2006, and Garrison 2007), we believe it is appropriate to act conservatively and apply our estimated mortality rate to all HMS shark gillnet effort.

Multiplying an immediate mortality rate of 24% by the total number of estimated sea turtle interactions annually results in the following estimates of immediate mortalities annually: 2.1 loggerheads, 0.7 Kemp's ridleys, 1.4 greens, 0.06 leatherback, and 0.06 hawksbill sea

³⁴ 2 of these 8 turtles had an unknown condition at catch and release, so we have conservatively estimated these as immediate mortalities.

turtles. Our estimates of lethal and non-lethal takes by species in Southeast shark gillnet gear, based on immediate mortality, are summarized in Table 5.14.

Table 5.14 Pre-Release Condition of Sea Turtles Estimated to Be Taken Annually in Southeast Shark Gillnet Gear

Species	Non-Lethal Take	Lethal Take Due to Immediate Mortality	Estimated Annual Take
Loggerhead	6.8	2.1	8.9 ^a
Kemp's ridley	2.3	0.7	3 ^a
Green	4.5	1.4	5.9
Leatherback	0.17	0.06	0.23
Hawksbill	0.17	0.06	0.23
Total	13.9	4.3	18.26

^a Estimated by Carlson and Richards 2011

Most, if not all, sea turtles released alive from gillnet gear will have experienced a physiological injury from forced submergence and/or traumatic injury from entanglement. Thus, in addition to the mortality observed at the time of release, some level of post-release mortality is expected for sea turtles released alive.

In August 2015, NMFS convened a workshop of experts to develop criteria for estimating post-interaction mortality³⁵ of sea turtles caught in trawl, gillnet, and trap fishing gear (Stacy et al. 2016). The results of the workshop were used in the development of national post-interaction mortality criteria and a criteria application process. Procedural instruction 02-110-21 provides guidance on the process for post-interaction mortality determinations of sea turtles bycaught in trawl, net, and pot/trap fisheries (NMFS 2017b). This directive reflects the most recent and best available information regarding post-interaction mortality for these gear types, and use of its criteria provides a mechanism to conservatively assess the potential impacts of the proposed action on sea turtles.

The criteria provided in the directive are based on the apparent degree of impairment, severity of physical injury, and relative risk of developing life-threatening conditions as a result of the interaction with gear. Sea turtles caught in fishing gear that are alive upon discovery exhibit a range of outward effects, from seemingly normal behavior and activity to complete unresponsiveness. Similarly, traumatic injuries of different degrees of severity are encountered, ranging from minor, superficial wounds to those that present an immediate threat to survival and risk of serious complications, such as secondary infections and diminished ability to forage and perform other vital biological functions. Because, in most instances, it is difficult to measure whether a sea turtle lived or died after being caught in fishing gear, the likelihood of mortality is best determined by activity level and the presence or absence of any abnormal behavior or injuries. There is inherent variability in the conditions under which observations are made and the amount of time sea turtles are available for examination due to factors such as fishing operations, sea state, weather, and time of day. In the criteria, each observation is categorized as low risk of mortality

³⁵ Post-interaction mortality is used interchangeably with post-release mortality in this Opinion.

(Category 1), intermediate risk of mortality (Category 2), or high risk of mortality (Category 3). Each mortality risk category is associated with percentages that reflect the proportion of sea turtles that are estimated to later die following release. In addition, injuries or conditions that are incompatible with survival are considered deaths (100% mortality). The mortality percentages applied to these risk categories were derived from a combination of expert opinion and available studies pertinent to sea turtle post-interaction mortality. Under the criteria, the lowest mortality risk category (Category 1) assigned for any interaction includes apparently uninjured sea turtles that exhibit indications of normal behavior and activity, those with slight alterations in behavior or activity that may still be considered within the bounds of normal, and turtles with minor, non-life threatening traumatic injuries. Category 1 has two estimated rates of post-interaction mortality, 10% (interactions at minimal risk of causing DCS), and 20% (interactions at risk of causing DCS). DCS concerns apply to sea turtles caught by sampling operating at a depth of 40 m or greater (NMFS 2017b). From 2007-2016, the average depth fished by the shark gillnet fishery was below the DCS threshold, at only 47 ft (14.3 m).

The SEFSC provided records of 33 sea turtles caught in Southeast shark gillnet gear from 2001-2009. These data remain the best available data on the condition of sea turtles that may be caught in the fishery. Although these data are primarily prior to many changes in the fishery, the gear is still expected to be fished in the same general manner as when there was more gillnet effort, and it is the manner in which the gear is fished (e.g., soak time) that affects the condition of the caught turtles. Other than the 8 already assigned immediate mortalities described above, 16 of the 33 turtles were caught uninjured and released alive and 9 were injured and released alive. Without knowing the specific injury to these turtles, we decided to take a conservative approach and assign the 9 injured turtles as Category 3 observed releases, which have a 0.80 post-release mortality rate. We assigned the 16 uninjured turtles a Category 1a mortality rate of 0.10 because the average depth of this fishery is less than 40 m. The average post-release mortality rate for these 25 turtles is 0.352 (SEFSC unpublished data) (9 turtles in Category 3 x 0.8 post-release mortality rate = 7.2 mortalities; 16 turtles in Category 1a x 0.1 post-release mortality rate = 1.6 mortalities. Total mortalities is 8.8 (7.2 + 1.6); 8.8 mortalities out of 25 turtles is 35.2 percent). Therefore, we decided to use 35% as the best available estimate for the post-release mortality of sea turtles in Southeast shark gillnet gear. The results of those calculations are in Table 5.15.

Table 5.15 Estimated Annual Sea Turtle Take, Immediate Mortalities, Post-Release Mortalities, and Non-Lethal Take in Southeast Shark Gillnet Gear

Species	Estimated Annual Take	Lethal Take Due to Immediate Mortality (24% of annual take)	Lethal Take Due to Post-Release Mortality (35% of released alive)	Total Estimated Annual Mortality
Loggerhead	8.9 ^a	2.1	2.4	4.5
Kemp's ridley	3 ^a	0.7	0.8	1.5
Green	5.9	1.4	1.6	3
Leatherback	0.23	0.06	0.06	0.12
Hawksbill	0.23	0.06	0.06	0.12
Total	18.26	3.9	4.9	9.2

^a Estimated by Carlson and Richards 2011

5.1.6.4 Estimated 3-Year Sea Turtle Takes and Mortalities in Southeast Shark Gillnet Gear

In the previous section, we concluded that 8.9 loggerhead sea turtles, 3 Kemp's ridley sea turtles, 5.9 green sea turtles, 0.23 leatherbacks, and 0.23 hawksbill sea turtles would be taken in Southeast shark gillnet gear annually. Thus, every 3 years we expect 26.7 loggerhead sea turtles, 9 Kemp's ridley sea turtles, 17.7 green sea turtles, 0.69 leatherback and 0.69 hawksbill sea turtles will be caught.

Of the turtles caught every 3-years, 13.5 (4.5×3) loggerheads, 4.5 (1.5×3) Kemp's ridleys, 9 (3×3) greens, 0.36 (0.12×3) leatherback, and 0.36 (0.12×3) hawksbill sea turtle takes are estimated to result in mortality. (Table 5.16).

Table 5.16 Estimated 3-Year Sea Turtle Takes and Mortalities in Southeast Shark Gillnet Gear

Species	Estimated 3-Year Take	Mortalities
Loggerhead	26.7	13.5
Kemp's Ridley	9	4.5
Green	17.7	9
Leatherback	0.69	0.36
Hawksbill	0.69	0.36
Total	54.8	27.7

5.1.7 Estimating Sea Turtle Incidental Catch in Smoothhound Gillnet Gear

In the following sections, we describe the approach used by Murray (2013) to estimate the number of loggerheads caught by smoothhound gillnet gear. These reports contain the best information available to determine the likely impacts of the smoothhound fishery on loggerhead sea turtles. This section also describes how we determined the number of non-loggerhead species caught by smoothhound gillnet gear, as well as our process for determining the number of lethal and non-lethal catch for all species. Murray (2013) includes a more detailed discussion of the data sources used, the calculation methods, the constraints of those methods, and the assumptions under which those calculations were made.

Estimating Loggerhead Sea Turtle Catch in Smoothhound Gillnet Gear

Murray (2013) estimated the total number of interactions between loggerhead and hard-shelled turtles and commercial gillnet gear in the Mid-Atlantic from 2007-2011 by using data collected by Northeast Fisheries Observer Program (NEFOP) observers and at-sea monitors (ASM). A generalized additive model (GAM) was used to estimate interaction rates (defined as the number of turtles per ton of fish landed), which were then applied to commercial Vessel Trip Report (VTR) data to estimate the total number of sea turtle

interactions. An annual average of 95 hard-shelled turtles were estimated to have interacted with gillnet gear in the Mid-Atlantic. Eighty-nine of those interactions were thought to be loggerheads (equivalent to 9 adults). Estimated rates of sea turtle bycatch and interactions have decreased compared to those from 1996-2006.

Murray (2013) estimated that from 2007-2011, 83 loggerhead sea turtles were caught by smoothhound gillnet gear in state and federal waters combined. The number of estimated loggerhead annual takes ranged from a low of 9 (in 2007) to a high of 26 (in 2011), with an annual average of 17. We have chosen to use the annual average of 17 for our estimate of annual loggerhead take by smoothhound gillnet gear because it is the best available estimate from more recent information. This is a decrease in bycatch compared to the 2002-2006 estimate of 159 loggerheads caught in this gear (with an annual average of 32 taken and a high of 53) as analyzed in the 2012 Opinion. Since this Opinion is analyzing the expected interactions of the operations in federal waters, below we estimate the percentage of those interactions occurring in federal waters.

Estimated Green, Kemp's Ridley, and Leatherback Sea Turtle Takes

Murray (2013) estimated the total take of non-loggerhead hard-shelled turtles in the smoothhound fishery. The estimated number of non-loggerhead takes ranged from a low of 0 (in 2007) to 2 (in 2009, 2010, and 2011), with an annual average of 2 from 2007-2011 (95% CI = 1-2; CV=0.38). These species are known to become entangled in gillnet gear, and the observed takes reported in Murray (2013) (other than hawksbills) are evidence of the presence and susceptibility of these species to gillnet gear in the Mid-Atlantic region. Because hawksbills were not observed entangled, we decided to assign one of these takes as a Kemp's ridley and one of these takes as a green sea turtle. Murray (2013) did not estimate takes of leatherbacks in the Atlantic gillnet fisheries, though they are also susceptible to gillnet gear. To calculate the number of leatherback sea turtle takes in the fishery, we followed a similar approach to the one described in Section 5.1.3.2, except we only used the percentages of sea turtles strandings from the Northeast because that is where the smoothhound fishery occurs.³⁶ Derived STSSN species abundance were 61.4% loggerheads, 15.9% green, 5.3% leatherbacks, 0.2% hawksbills and 17.2% Kemp's ridleys. As above, the total number of all sea turtles (N_{Total}) taken annually was calculated by dividing the Murray (2013) annual estimate of loggerheads taken in the smoothhound fishery (17) ($N_{BLL Lo}$) by the loggerhead species abundance ($STSSN_{Lo}$) (61.4 percent). The number of leatherback ($N_{BLL Le}$) was estimated by multiplying the species abundance ($STSSN_{Le}$) by the total number of sea turtles taken annually (N_{Total}). We have summarized our estimates in Table 5.17.

Murray (2013) estimated the loggerhead take for both state and federal waters. The gears used to target smoothhound in federal and state waters are the same. The time of year when the fishery operates is also generally the same across state and federal waters. The species of sea turtles that occur in the action area are all highly migratory and found in both state and federal waters. The vast majority of both state and federal fishing effort likely occurs in the depth range 0-120 ft., where sea turtles are known to occur most frequently. Since the gear, timing, and distribution of effort with respect to sea turtle abundance are

³⁶ 1) $N_{GN Lo} = STSSN_{Lo} \times N_{Total}$; 2) $N_{GN Lo} \times STSSN_{Lo}^{-1} = N_{Total}$; 2) $N_{GN Le} = STSSN_{Le} \times N_{Total}$

essentially the same in both state and federal waters, both the state and the federal fishery are likely to have a similar rates of entanglement of sea turtles. VTR data often includes information on the distance from shore where smoothhound were caught.³⁷ Since the subject of the consultation is smoothhound fishing in federal waters, we used the distance from shore information provided by the VTR data to estimate smoothhound fishing effort in federal waters. From 2004-2011, the proportion of smoothhound landings coming from federal waters was the highest in 2004 at 47%; the lowest in 2007, 27%; and the mean was 36% (VTR Database unpublished data). We acted conservatively and used 47% in our calculations to estimate smoothhound effort in federal waters. Applying that 47% fishing effort rate to our take calculations yielded an estimate of the annual number of loggerheads likely caught during smoothhound fishing in federal waters. To act conservatively we will assume the Kemp's ridley, green, and leatherback sea turtle takes all occurred in federal waters. Table 5.17 displays the estimated number of annual takes in federal waters for each species.

Table 5.17 Estimated Annual Sea Turtle Incidental Take in Smoothhound Gillnet Gear

Species	Estimated Annual Take in State and Federal Waters	Total Estimated Annual Take in EEZ
Loggerhead	17	8
Kemp's ridley	1	1
Green	1	1
Leatherback	1.5	1.5
Total	20.5	11.5

5.1.7.2 Estimating Immediate and Post-Release Mortality of Sea Turtles in Federal Smoothhound Gillnet Gear

The ultimate fate of animals incidentally caught is needed to conduct an effective jeopardy analysis. Murray (2013) estimated 58% (52 loggerheads equivalent to 5 adults) of loggerhead interactions were considered to result in mortality. The 58% estimate was based on Upite et al. (2013) and did not differentiate between immediate and post-release mortality, turtle life stages, or species, but is the best available scientific information. Using this estimate of 58%, we calculated the following annual mortality levels for sea turtles caught in smoothhound gillnet gear: 4.6 loggerheads, 0.6 Kemp's ridleys, 0.6 greens, and 0.9 leatherbacks. Our total estimated sea turtle incidental catch and annual mortality in smoothhound gillnet gear are summarized in Table 5.18.

Table 5.18 Estimated Annual Sea Turtle Incidental Take and Mortalities in Smoothhound Gillnet Gear

Species	Total Annual Estimated Take	Mortalities
Loggerhead	8	4.6
Kemp's ridley	1	0.6
Green	1	0.6

³⁷ Distance from shore categories for the Atlantic include: inland, inshore (0-3 miles), EEZ (3-200 miles), and international (200+ miles).

Leatherback	1.5	0.9
Total	11.5	6.7

5.1.7.3 Estimated 3-Year Sea Turtle Takes and Mortalities by Smoothhound Gillnet Gear

In the previous section, we concluded that 8 loggerhead, 1 Kemp's ridley, 1 green, and 1.5 leatherback sea turtles would be caught annually by smoothhound gillnet gear. Thus, every 3 years we expect 24 loggerheads, 3 Kemp's ridleys, 3 greens, and 4.5 leatherbacks would be caught by smoothhound gillnet gear. Of the turtles caught every 3 years, 13.8 (4.6*3) loggerheads, 1.8 (0.6*3) Kemp's ridleys, 1.8 (0.6*3) greens, and 2.7 (0.9*3) leatherback sea turtle takes are estimated to result in mortality (Table 5.19).

Table 5.19 Estimated 3-Year Sea Turtle Takes and Mortalities in Smoothhound Gillnet Gear

Species	Estimated 3-Year Take	Mortalities
Loggerhead	24	13.8
Kemp's Ridley	3	1.8
Green	3	1.8
Leatherback	4.5	2.7
Total	34.5	20.1

5.1.8 Vessel Interactions with Sea Turtles

HMS vessels transiting to and from fishing areas and moving during fishing activity pose a potential threat to sea turtles. Based on recorded sizes of stranded sea turtles with propeller injuries, both juvenile and adult sea turtles are subject to vessel strikes. Young sea turtles are very alert and so less likely to be hit by a vessel. Sea turtles are susceptible to vessel collisions and propeller strikes because they regularly surface to breathe and may spend a considerable amount of time on or near the surface of the water. Activities such as basking, mating, and resting at the surface also make these animals susceptible to vessel strikes. For example, Sobin (2008) suggests loggerhead sea turtles are most vulnerable to boat strikes following a false crawl event, within 12 hours after nesting, and the night before returning to the beach to nest, when they are closest to shore and also subject to high-traffic boat areas. Sea turtle stranding data also indicates sea turtle species may be more susceptible to being hit by boat propellers during movements associated with reproductive activity (Foley et al. 2008b). Sick and injured sea turtles typically float so are also particularly vulnerable to being struck by vessels.

5.1.8.1 Types of Interactions (Stressors and Individual Responses to Stressors if Exposed)

Vessel strikes may result in direct injury or death through collision (concussive) impacts or propeller wounds. Although sea turtles, with the exception of leatherback sea turtles, have hard carapaces, they are unable to withstand the strike of a rapidly moving vessel or the cut

of a propeller. A sea turtle's spine and ribs are fused to the shell, which is a living part of their body that grows, sheds, and bleeds. Rapidly moving vessels may strike the head or carapace and result in fractures. Injuries to the carapace can involve fractures to the spinal column and buoyancy problems. A propeller can easily cut through the shell and sever or damage the spine and internal organs. Propeller injuries may range from mild to severe and include head lacerations, eye injury, injury to limbs, and carapace lacerations and fractures. Chronic and/or partially healed propeller wounds also may be associated with secondary problems such as emaciation and increased buoyancy (Jacobson et al. 1989). Abnormally buoyant sea turtles are unable to dive for food or escape predators or future vessel strikes. Seriously injured or dead turtles may be struck multiple times by vessels before they drift ashore.

The proportion of vessel-struck sea turtles that survive or die is unknown. In many cases, it is not possible to determine whether documented injuries on stranded animals resulted in death or were post-mortem injuries. Sea turtles found alive with concussive or propeller injuries are frequently brought to rehabilitation facilities; some are later released and others are deemed unfit to return to the wild and remain in captivity. Sea turtles in the wild are documented with healed injuries; thus, we know at least some sea turtles survive without human intervention.

5.1.8.2 Potential Factors Affecting the Likelihood and Frequency of Sea Turtle Exposure to Vessel Strikes

The threat posed by moving vessels is not constant and is influenced in part by vessel type (planing versus displacement hulls), vessel speed, and environmental conditions such as sea state and visibility. Seasonal and regional variance in vessel use and sea turtle distribution and densities also are expected to affect sea turtle vessel strike rates. Below we review how these factors may affect the likelihood and frequency of sea turtle vessel strikes.

Vessel Type and Speed

Generally, vessels typically possess either a planing hull or a semi-displacement hull. Planing hulls, typical of smaller (e.g., 18-27 ft in length) recreational vessels, are designed to run on top of the water (i.e., on plane) at high speeds. Conversely, displacement hulls push through the water, as they have no hydrodynamic lift, and the boat does not rise out of the water as speed increases. Because of how these two hulls function, they likely introduce differing threat risks to sea turtles. For example, because operational speeds of planing hulls are typically greater than displacement hulls, they possess greater kinetic energy to transfer to an impacted sea turtle. Additionally, because most of the hull is out of the water, the running gear (including the propeller and skeg of an outboard) of a planing hull running at speed becomes a significant cutting/slashing threat, in combination with the concussive effect of a collision. This risk would be compounded by twin or triple engines, which are fairly common in small- to medium-sized (e.g., 25-34 ft in length) recreational HMS. In comparison, displacement hulls, which include most large (e.g., > 65 ft in length) vessels comprising commercial traffic (e.g., tankers, freighters, tugs, etc.), while traveling slower extend deeper into the water column. The slower speed and greater size of these vessels suggests the risk to sea turtles is largely limited to a concussive impact from the

hull. It is possible that a sea turtle may avoid significant impact altogether by being pushed away by the hydrodynamic bow wave of a large vessel, and, therefore, allowed to escape before incurring an injury.

Greater vessel speed is expected to increase the probability that a sea turtle would fail to have time to flee the approaching vessel and that the vessel operator would fail to detect and avoid the sea turtle. A study on vessel speed and collisions with green sea turtles conducted in shallow water (<5 m) along the northeastern margin of Moreton Bay, Queensland, Australia, analyzed behavioral responses of benthic green sea turtles to an approaching 20-ft (6-m) aluminum vessel at slow (2 knot), moderate (6 knot), and fast (10 knot) speeds (Hazel et al. 2007). The proportion of turtles that fled to avoid the vessel decreased significantly as vessel speed increased, and turtles that fled from moderate and fast approaches did so at significantly shorter distances from the vessel than turtles that fled at slow approaches. Although vessel noise is within a green turtle's hearing range, there are several factors that may impede their recognition of the noise as a threat (e.g., directionality of the noise in the ocean and habituation to background vessel noise). The results implied that vessel operators could not rely on sea turtles to actively avoid being struck by a vessel if it exceeds 2 knots. On this basis, the authors determined that vessel speed was a significant factor in the likelihood of a strike and implied that mandatory vessel speed restrictions were necessary to reduce the risk of vessel strikes to sea turtles (Hazel et al. 2007).

Environmental Factors

Sea state and visibility will also influence the likelihood of an interaction between a vessel and a sea turtle. Typically, most vessel operators keep watch for potential obstructions or debris, which can seriously damage or potentially sink a boat. The calmer the sea state, the easier it is to see floating objects, including sea turtles. When the sea state increases and swells are introduced, observing floating obstructions gets increasingly difficult. However, increased sea state will also compel most vessels on the water to decrease speed, which would reduce the risk of a strike and potentially the severity of a strike. Also, generally fewer recreational vessels go on trips in rough conditions, in comparison with calm seas. Thus, there may be a seasonal component to the magnitude of vessel strike risks to sea turtles in some areas. Another factor is traveling east or west during a rising or setting sun; this can dramatically limit forward visibility and inhibit an operator from avoiding a floating sea turtle or other obstruction.

Vessel Traffic and Sea Turtle Abundance

Areas with high concentrations of vessel traffic and high concentrations of sea turtles are expected to have a higher probability and frequency of vessel strikes than areas where vessels and/or sea turtles are less abundant. Data on offshore vessel traffic is still largely absent, but several recent studies have explored the issue of vessel traffic for a few coastal counties in Florida (Sidman et al. 2007; Sidman et al. 2005; Sidman et al. 2009). The available information indicates that there is extensive traffic in inshore and nearshore waters, particularly around inlets. Additionally, there are latitudinal changes in peak use and average number of trips, with a longer peak season and higher number of monthly trips in southern counties when compared to northern counties.

5.1.8.3 Estimating Sea Turtle Vessel Strikes Attributable to HMS Vessels

It is very difficult to definitively or even approximately evaluate the potential risk to sea turtles stemming from specific vessel traffic from any action because of the numerous variables discussed in Section 5.1.8.2 that may impact vessel strike rates. This difficulty is compounded by a general lack of information on vessel use trends, particularly in regard to offshore vessel traffic. Available data are insufficient to account for such differences in our analysis. However, the following analysis is intended to provide a gross estimate of the potential impact HMS vessels may have on sea turtles, taking a reasoned approach to conservatively account for vessel impacts based on the best available information.

Foley et al. (2008b) evaluated distributions, relative abundances, and mortality factors, including vessel strikes, for sea turtles in Florida from 1980 through 2005 as determined from strandings and Foley et al (2017) recently updated this information. These analyses remain the best available comprehensive quantitative evaluations of vessel strike impacts to date. The Florida Sea Turtle Stranding and Salvage Network (FLSTSSN) has documented 36,425 Florida stranding records (all species and size classes) in their database from 1980 through 2014 (Foley et al. 2008b). Vessel-strike injury (VSI) was the most commonly noted external anomaly for stranded sea turtles in Florida that indicated a potential mortality factor. From 1980 through 2014, there were 7,509 sea turtle stranding records in Florida with a VSI (2,718 green, 142 leatherback, 4,196 loggerhead, 401 Kemp's ridley, and 52 hawksbill sea turtles). By species, the % occurrence of a VSI was 36% green, 2% leatherback, 56% loggerhead, 5% Kemp's ridley, and 1% hawksbill sea turtles. Based on the STSSN strandings data, there was an average of 220 sea turtles injured or killed per year due to VSI (7,509 sea turtles/34 years) (Foley et al. 2017). This is likely an under-representation because other strandings may have had a VSI but were not coded for this and because some turtles with VSI did not strand. The numbers would increase by approximately 50% (or 330 sea turtles/year) if all the injuries possibly related to vessel-strikes were included; however, they had been coded for other stranding reasons.

In a January 12, 2009, memorandum from Michael Barnette, SERO fishery biologist, to David Bernhart, SERO Assistant Regional Administrator for Protected Resources, the potential threats on listed sea turtles from vessel traffic related to new dock and/or marina construction in Florida were analyzed. In doing so, several different estimates of vessel strike frequency on a by-vessel and by-trip basis with varying degrees of conservatism were presented by using Foley et al. (2008b)'s analysis of Florida sea turtle stranding data attributed to vessel impacts, from 1980 to 2005, discussed above, in combination with Florida vessel traffic and use trend data under various assumptions. The number of injured or killed sea turtles attributed to vessel strikes was estimated using various assumptions. Under a less conservative approach, it was assumed that those strandings where the turtles were stranded alive with propeller injuries, where the turtles were determined to have been hit before death, or where the turtles were found freshly dead with propeller wounds were killed by vessel strikes. Thus, 1,086 sea turtles over a 25-year period met these criteria, and thus under these assumptions, 43 sea turtles were assumed to have been injured or killed by vessel strikes a year. Another, more conservative estimate was that all 3,586 stranding records with propeller injuries (not just the 1,086 discussed above) and the 703 stranding records with crushing injuries were pre-mortem and caused by vessels (i.e., 4,289

total potential vessel related sea turtle injuries so 171 sea turtles injured or killed a year [4,289 sea turtles / 25 years]). The minimum and maximum total number of potential vessel trips in Florida waters during the course of a year was estimated based the number of registered vessels in Florida coastal counties in 2007 and an extrapolation of the minimum and maximum average number of trips per vessel per month documented by several Florida county recreational vessel traffic studies (Sidman et al. 2005 and 2007). The total number of potential vessel trips in Florida ranged from 25.6 to 53.1 million trips. Assuming each vessel trip possesses the same likelihood of resulting in a sea turtle strike, based on the best available information, Barnette estimated a sea turtle vessel strike was to occur: (1) every 1,235,268 trips under the least conservative approach (43 vessel strikes per year with 53.1 million trips per year), (2) every 149,877 trips under a more conservative approach (171 vessel strikes per year with 25.6 million trips per year), and (3) every 10,491 to 19,490 trips under the “ultra-conservative” approach. Under this latter approach, based on a study of beach strandings in North Carolina as an indicator of at sea-mortality in an offshore commercial fishery (Epperly et al. (1996)), Barnette assumed that the 171 strandings per year represented only 7-13% of the total mortalities (such that there were 1,315 to 2,443 sea turtle mortalities a year as a result of vessel interactions). Under the low effort scenario (25.6 million trips a year), this would result in a strike every 10,491 to 19,490 trips.

On April 18, 2013, Barnette updated the January 12, 2009, threats and effects analysis memorandum, but the information did not significantly change from the 2009 memorandum. The estimates of the number of trips per sea turtle vessel strike under the different scenarios remained the same thus are still the best available information. NMFS will be updating the 2013 memo based on Foley et al. (2017).

In order to roughly gauge the potential impacts of vessel interactions on sea turtles, we very conservatively assumed all HMS trips also possess the same likelihood of resulting in a sea turtle strike and applied the vessel strike trip rates from the Barnette memorandum. We used annual average number of trips from each fishery sector and then summed them for our future effort proxy for all sectors combined. In Table 5.20, we present the 2012-2015 average number of trips from each sector and their sum them for our future effort proxy for all HMS fisheries.

Table 5.20 2011-2015 Average Number of HMS Vessel Trips in the Atlantic Region and GOM EEZ by Sector and All Sectors Combined

Vessel trips	Commercial	Private	Charter	All Sectors Combined
2011-2015 Average	1,667	102,104	20,557	124,328

Source: SEFSC's Commercial Logbook Data Program unpublished data; April 2017; NMFS Fisheries Statistics Division, July 2017. Caribbean recreational trip data only available for 2014 and 2015.

If there are 124,328 annual trips in HMS fisheries, based on Barnette’s “ultra-conservative” approach, 6.38 to 11.85 sea turtles would be hit annually ($124,328/19,490 = 6.38$; $124,328/10,491 = 11.85$). Based on the 2011-2015 average number of total trips in the fishery and the above vessel strike rates, estimated vessel strikes attributed to HMS

fisheries could range from none, under the least conservative approach (124,328/1,235,268=0.1), up to 7 to 12 sea turtles under the most conservative approach.

Barnette did not consider his most conservative approach to be a realistic estimate for considering the potential vessel impact risk associated with typical dock and/or marine construction. He stated that due to the long string of extrapolations, estimates, and assumptions, as well as some other inherent issues with basing conclusions on Florida recreational vessel traffic patterns (i.e., largely nearshore/coastal) with a single, limited study conducted on mortalities in a North Carolina commercial fishery, his most conservative approach was intended solely to help define the worst-case scenario. In addition, the assumption that all stranded turtles with vessel strike impacts were struck pre-mortem likely overestimates the number of reported strandings attributed to vessels. This is because, although it is highly likely that more than 13% of records were pre-mortem and directly attributed to being vessel-struck, it is equally likely that at least some sea turtles struck were dead from other causes prior to being struck. Thus, to try and balance these considerations, we believe using our lower estimate of the most conservative method (i.e., 6.38, rounded to 7 sea turtle strikes annually or 21 every 3 years) is conservative enough to satisfy our intent to be conservative and err on the side of the species. Based on the % occurrence of strandings with propeller wounds by species from Foley et al. (2017) (i.e., 56% loggerhead, 36% green, 5% Kemp's ridley, 2% leatherback, and 1% hawksbill), we estimate that of the 21 sea turtles that may be struck and killed every 3 years, 11.8 may be loggerhead sea turtles, 7.6 may be green sea turtles, 1.1 may be Kemp's ridley sea turtles, 0.4 may be a leatherback sea turtle, and 0.2 may be a hawksbill sea turtle.

In reality, this crude assumption likely exaggerates the risk of vessel strikes the HMS fisheries poses to sea turtles, given what we know about potential factors affecting the likelihood and frequency of sea turtle exposure to vessel strikes. For example, vessel strike rates off Florida are likely much higher given Florida waters typically have greater amounts of both fishing vessels and sea turtles. However, with the limited available information, we believe, while imprecise, this provides a conservative, reasoned approach to recognize and account for some potential vessel strike impacts attributed to the fishery's vessels.

5.1.9 Summary of Estimated Sea Turtle Fishery Catch, Mortalities, and Vessel Strikes from the Proposed Action

In Table 5.21 we present 3-year estimated takes and mortalities we anticipate under the proposed action based on the analyses we presented in the preceding sections. We chose to present all of the estimates in this manner primarily to help standardize our sea turtle catch estimates, but also to be consistent with the 3-year approach used in our ITS. By presenting the data in 3-year estimates, we are able to consider all of the cumulative takes over time more easily. In addition, our annual catch estimates are based on averages, so the numbers of annual catch are likely to fluctuate above and below the number specified from year to year. Thus, we decided to consider all of our take estimates in 3-year periods to incorporate annual variability. We conservatively rounded the 3-year take and mortality estimates up to the nearest whole number.

North Atlantic and South Atlantic Green Sea Turtle DPSs

As described in Section 3.3.5, information suggests that the vast majority of the anticipated green sea turtles caught in the Gulf of Mexico and South Atlantic regions are likely to come from the North Atlantic DPS. However, it is possible that animals from the South Atlantic DPS could be captured during the proposed action. Because the proposed action occurs in the South Atlantic, the Gulf of Mexico, and the Caribbean, we assume based on Bass and Witzell (2000) that 95% of animals captured during the proposed action are from the North Atlantic DPS. Our analysis of the South Atlantic DPS will consider that 5% of the green sea turtles affected by the proposed action are from the South Atlantic DPS. Applying these percentages to our estimated takes of 48 green sea turtles (rounded up) (26 lethal and 22 nonlethal) every 3 years and rounding in such a way as to conservatively assume the most lethal captures, results in an estimated catch of up to 46 green sea turtles from the North Atlantic DPS ($48 \times 0.95 = 45.6$, rounded up), of which 25 are expected to be lethal ($26 \text{ rounded up} \times 0.95 = 24.7$, rounded up) and 21 are expected to be nonlethal ($22 \text{ rounded up} \times 0.95 = 20.9$, rounded up) and an estimated catch of up to 3 green sea turtles from the South Atlantic DPS ($48 \times 0.05 = 2.4$, rounded up), of which 2 are expected to be lethal ($26 \times 0.05 = 1.3$ rounded up) and, therefore; 1 is expected to be nonlethal. We note rounding when splitting the take into the two DPSs results in a slightly higher combined total than the 3-year actual estimate (i.e., 49 instead of 48).

Table 5.21 Summary of Estimated 3-Year Sea Turtle Take (T) and Mortality (M) Estimates by the Proposed Action

	Loggerhead		Kemp's ridley		Green		Leatherback		Hawksbill	
	T	M	T	M	T	M	T	M	T	M
Shark Bottom Longline	23.7	10.5	7.2	3	15.9	6.9	0.6	0.4	0.6	0.4
HMS Vertical Line	4.5	0.9	1.4	0.3	3	0.6	0.1	0.02	0.1	0.02
Southeast Shark Gillnet	26.7	13.5	9	4.5	17.7	9	0.69	0.36	0.69	0.36
Smoothhound Gillnet	24	13.8	3	1.8	3	1.8	4.5	2.7	0	0
All Gear Impacts Combined	78.9	38.7	20.6	9.6	39.6	18.3	5.9	3.5	1.4	0.8
Vessel Strike	11.8	11.8	1.1	1.1	7.6	7.6	0.4	0.4	0.2	0.2
Total	90.7	50.5	21.7	10.7	47.2	25.9	6.3	3.9	1.6	1
Total Rounded Up	91	51	22	11	46*	25*	7	4	2	1
					3**	2**				

*North Atlantic DPS of green sea turtle

**South Atlantic DPS of green sea turtle

We estimate the total take in the HMS fisheries subject to this consultation would be no more than 171 sea turtles, comprised of 91 loggerheads, 22 Kemp's ridley, 46 green North Atlantic DPS, 3 green South Atlantic DPS, 7 leatherbacks, and 2 hawksbills sea turtles caught every 3 years. Applying our overall mortality rates and conservatively rounding up the final numbers, we estimate that up to 51 loggerhead sea turtles, 11 Kemp's ridley sea

turtles, 25 green North Atlantic DPS, 2 green South Atlantic DPS, 4 leatherback sea turtles, and 1 hawksbill sea turtles may be killed, every 3 years.

5.2 Effects on Smalltooth Sawfish

Of the gear types used in the HMS fisheries by commercial and/or recreational fishers (i.e., hook-and-line gear, gillnets, harpoon, speargun, and purse seines), we believe bottom longline, rod and reel, and gillnet gear may affect and are likely to adversely affect smalltooth sawfish. This section focuses on evaluating the effects of HMS hook-and-line gear (namely, bottom longline and rod and reel) and gillnet gear on smalltooth sawfish.

5.2.1 Types of Interactions with Smalltooth Sawfish and Hook-and-Line Gear and Gillnet Gear

Hook-and-line gear and gillnet gear are known to adversely affect smalltooth sawfish via hooking and/or entanglement. Hooking and entanglement can lead to cuts, puncture wounds, or lost rostral teeth. Hooked or entangled smalltooth sawfish may potentially also suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns, though we have no actual evidence of such effects. Observer data indicate that regardless of the type of interaction, the vast majority of incidentally caught smalltooth sawfish are released alive and in good condition. The following discussion summarizes in greater detail the available information on how individual smalltooth sawfish may respond to interactions with hook-and-line gear and gillnet gear.

Hooking

Based on commercial observer data, data from Mote Marine Laboratory bottom longline research surveys, and from reported recreational rod-and-reel fishing encounters, the vast majority of smalltooth sawfish are hooked in the mouth (ISED 2014). Foul-hooking reports are not nearly as frequent, but they do occasionally occur. There is only a single report of a smalltooth sawfish deeply hooked (ISED May 2009). Once hooked, the gangion or leader frequently gets wrapped around the animal's saw. This may result from slashing during the fight, spinning on the line as it is retrieved, or any other action bringing the rostrum in contact with the line.

Based on available data, all smalltooth sawfish caught on vertical lines and most smalltooth sawfish caught on bottom longline gear survive. Between 2008 and 2015, 25 smalltooth sawfish ranging in size from 2.4 m to 4.6 m (7.9 ft to 15 ft) were observed caught in shark bottom longline gear in the Atlantic and Gulf of Mexico. All of the animals were released alive except one. Soak times do not seem to be a factor for smalltooth sawfish mortality. It has been hypothesized that because the animal's natural habit consists of lying on the sea floor and using its spiracles to breathe that survivorship should be high (Simpfendorfer et al. 2010b). Thorson (1982) reported that largetooth sawfish caught by fishers at night or when no one was present to tag them were left tethered in the water with a line tied around the rostrum for several hours with no apparent harmful effects. Additional information on survivorship of smalltooth sawfish comes from Mote Marine Laboratory research using bottom longline, nets, and rod and reel on the Southwest coast of Florida. From 2000-

2008, over 130 individuals ranging in size from 62 cm to 496 cm (24 in to 195 in) were caught, 21 of which were caught on bottom longline gear. All of these individuals were alive upon catch and safely released with no apparent harm to the fish (T. Wiley-Lescher, Haven Worth Consulting, pers. comm. to S. Norton, NMFS, July 2013).

There are no studies on the post-release mortality of smalltooth sawfish. Based on tag-recapture data, post-release mortality is expected to be low. Still, sublethal effects on smalltooth sawfish may occur, particularly if the animal is removed from the water. The weight of the sawfish on dry land (or aboard a vessel) may damage internal organs; moreover, the stress of being removed from the water may also cause sublethal effects.

Entanglement

Smalltooth sawfish are particularly vulnerable to entanglement in gillnets. Early publications document their frequent capture in this gear type and gillnets are believed to be one of the primary causes for the species' decline. As previously mentioned in Section 3.2.9, the long, toothed rostrum of the smalltooth sawfish easily penetrates netting, causing entanglement when the animal attempts to escape. The monofilament mesh can inflict abrasions and cuts, cause bleeding, and hinder feeding behavior. Even a few strands of monofilament can cause significant damage (C. Simpfendorfer pers. comm.).

The toothed rostrum also makes it very difficult to disentangle a smalltooth sawfish without harming the animal. Entangled animals frequently have to be cut free, causing extensive damage to nets. The entangled smalltooth sawfish can also endanger fishermen if brought onboard a vessel. For these reasons, many historical records of smalltooth sawfish catches note they were either killed or released after their saws had been removed (e.g., Henshall 1895, Evermann and Bean 1897, Bigelow and Schroeder 1953).

Effects on smalltooth sawfish from incidental capture in gillnets today likely depend on fishermen's handling practices. For example: (1) the amount of gear and time fishermen are willing to sacrifice to carefully remove an animal; (2) whether the animal is restrained while being handled to avoid damage to the rostrum and rostral teeth; (3) the length of time an animal is out of the water while being disentangled; and (4) the amount of gear left on the animal when released, are all likely to impact the overall severity of the event.

5.2.2 Factors Affecting the Likelihood of Smalltooth Sawfish Exposure to Hook-and-Line Gear

A variety of factors may affect the likelihood of smalltooth sawfish interactions with hook-and-line gear. The spatial overlap between fishing effort and smalltooth sawfish abundance is the most noteworthy variable involved in anticipating interactions. Other important factors for determining the likelihood and frequency of interactions include the types of gear used (e.g., baits, hooks) and the fishing techniques employed.

Spatial/Temporal Overlap between Fishing Effort and Smalltooth Sawfish

The spatial distribution of smalltooth sawfish influences the rate of interaction with fishing gears. The more abundant smalltooth sawfish are in a given area where fishing occurs, the greater the probability a sawfish will interact with gear. The temporal distribution of fishing effort and smalltooth sawfish abundance is also a factor.

Different life stages of smalltooth sawfish are associated with different habitat types and water depths. Very small and small juvenile smalltooth sawfish are most commonly associated with shallow water areas of Florida, close to shore and typically associated with mangroves (Simpfendorfer and Wiley 2004a). Since larger (> 200 cm in length) size classes of the species are also observed in very shallow waters, it is believed that smaller (younger) animals are restricted to shallow waters, while larger animals roam over a much larger depth range (Simpfendorfer 2001). Poulakis and Seitz (2004b) observed that nearly half of the encounters with adult-sized sawfish in Florida Bay and the Florida Keys occurred in depths from 200-400 ft. (70-122 m). Simpfendorfer and Wiley (2005b) also reported encounters in deeper water off the Florida Keys, noting that these were mostly reported during winter. Observations on commercial longline fishing vessels and fishery independent sampling in the Florida Straits report large sawfish in depths up to 130 ft. (~ 40 m) (J. Carlson, NMFS SEFSC and G. Burgess, FMNH pers. comm.).

Large juveniles and adult smalltooth sawfish are known to occur in water depths of 100 m or more. Thus, HMS hook-and-line gears deployed in deeper water are more likely to encounter these two size classes.

Soak Time/Number of Hooks

Bottom longline gear interactions with smalltooth sawfish may be influenced by both soak time and the number of hooks fished. The longer the soak time, the longer a smalltooth sawfish may be exposed to an entanglement or hooking threat, increasing the likelihood of such an event occurring. Likewise, as the number of hooks fished increases, so does the likelihood of an incidental hooking event.

Hook Type

The type of hook (size and shape) may impact the probability and severity of interactions with smalltooth sawfish. The point of a circle hook is turned toward the shank, while the point of a J-hook is not. Thus, the configuration of a circle hook may reduce the likelihood of foul-hooking interactions because the point of the hook is less likely to accidentally become embedded in the smalltooth sawfish's mouth. Circle hooks make gut-hookings unlikely. Such interactions are believed to be extremely rare and there is only a single known record of such despite hook-and-lines being the most common source of encounter records.

Bait

Smalltooth sawfish feed primarily on fish and crustaceans. Mullet, jacks, and ladyfish are believed to be their primary food sources (Simpfendorfer 2001). Smalltooth sawfish are reported to subsist on schooling fish such as mullet and clupeids (74 FR 45353, September 2, 2009). There is currently no data available on the attraction of smalltooth sawfish to bait used in the shark bottom longline fishery; however, as sawfish are caught on bottom

longlines at least some of the baits used in these fisheries appear to attract smalltooth sawfish.

Environmental Conditions

Environmental conditions may also play a part in whether a smalltooth sawfish interacts with hook-and-line gear. Fishing gear can drift according to oceanographic conditions, including wind and waves, surface and subsurface currents, etc.; therefore, depending on these species' behavior, environmental conditions, and location of the set, smalltooth sawfish can become entangled in the gear.

5.2.3 Estimating Smalltooth Sawfish Incidental Catch in Shark Bottom Longline Gear

The SEFSC estimated the level of protected resource take from 2008-2015 in shark bottom longline gear (Carlson et al. 2017). In the following sections, we describe the take estimates calculated in that report. Carlson et al. (2017) include more detailed discussion of the data sources used, calculation methods, constraints of those methods, and the assumptions under which those calculations were made.

5.2.3.1 Observer Data Summary

In total, 25 smalltooth sawfish were observed caught in the Gulf of Mexico and South Atlantic regions from 2008-2015 (Carlson et al. 2017) (Table 5.22).

Table 5.22 Observed Catch of Smalltooth Sawfish by Region, 2008-2015: Gulf of Mexico (GOM) or South Atlantic (SA)

Fishery	Year	Quarter	Area	Condition
SHX BLL - Non-Research Fishery	2008	3	GOM	Alive
SHX BLL - Non-Research Fishery	2008	4	GOM	Alive
SHX BLL - Research Fishery	2009	1	GOM	Alive
SHX BLL - Research Fishery	2009	1	GOM	Alive
SHX BLL - Research Fishery	2009	2	GOM	Alive
SHX BLL - Research Fishery	2009	2	GOM	Alive
SHX BLL - Research Fishery	2009	2	GOM	Alive
SHX BLL - Research Fishery	2010	2	GOM	Alive
SHX BLL - Research Fishery	2010	2	GOM	Alive
SHX BLL - Research Fishery	2010	2	GOM	Alive
SHX BLL - Research Fishery	2010	3	GOM	Alive
SHX BLL - Non-Research Fishery	2010	3	GOM	Alive
SHX BLL - Research Fishery	2010	4	GOM	Alive
SHX BLL - Research Fishery	2011	2	GOM	Alive
Fishery	Year	Quarter	Area	Condition
SHX BLL - Research Fishery	2011	3	GOM	Alive
SHX BLL - Research Fishery	2012	4	SA	Dead
SHX BLL - Research Fishery	2013	2	GOM	Alive
SHX BLL - Research Fishery	2013	2	GOM	Alive
SHX BLL - Research Fishery	2014	3	SA	Alive

SHX BLL - Research Fishery	2014	4	SA	Alive
SHX BLL - Research Fishery	2014	4	SA	Alive
SHX BLL - Research Fishery	2014	4	SA	Alive
SHX BLL - Research Fishery	2014	4	GOM	Alive
SHX BLL - Research Fishery	2015	2	GOM	Alive
SHX BLL - Research Fishery	2015	2	GOM	Alive

Source: Carlson et al. 2017

5.2.3.2 Extrapolated and Observed Smalltooth Sawfish Catch in Shark Bottom Longline Gear

Carlson et al. (2017) extrapolated the total number of smalltooth sawfish caught in the non-research shark fishery in the Gulf of Mexico and South Atlantic from 2008-2015 was 28.8. An additional 22 smalltooth sawfish were observed caught in shark research fishery during the same period; primarily from the Gulf of Mexico (Carlson and Richards 2011, Carlson et al. 2017, NMFS unpublished data). In total, an estimated 50.8 smalltooth sawfish were caught by shark bottom longline gear in the Gulf of Mexico and South Atlantic from 2008-2015 (Table 5.23).

Table 5.23 Extrapolated and Observed Smalltooth Sawfish Catch in Shark Bottom Longline Gear

Fishery	Year	Total Take**
Research*	2008	0
Research	2009	5
Research	2010	5
Research	2011	2
Research	2012	1
Research	2013	2
Research	2014	5
Research	2015	2
Non-research (Extrapolated)	2008-2015	28.8
Total Extrapolated and Observed (2008-2015)		50.8
Annual Average (2008-2015)		6.4
*Research takes are all observed.		
**Carlson et al. (2017) also provides the 95% CIs and CVs for the estimated catch in the non-research shark fishery		

Source: Carlson and Richards 2011 et al. 2017, NMFS unpublished data)

Discussion of Assumptions and Factors Influencing Accuracy of Extrapolated Take Estimate

The small sample size of observed incidental catch in the non-research shark fishery constrained the extrapolation of fishery-wide take estimates. The rarity of incidental catch events is a problem because estimates are based on only one or a few events. Additionally, sparse data may not fit a critical assumption of the delta lognormal model (Pennington 1983) that the non-zero CPUEs are drawn from a lognormal distribution (Carlson and Richards 2011). Nonetheless, with the current levels of observer coverage, these estimates represent the best available information regarding smalltooth sawfish interactions with the

fleet and provide the best picture of the likely interactions that occurred between 2008 and 2015.

5.2.3.3 Estimating Smalltooth Sawfish Mortality in Shark Bottom Longline Gear

Carlson et al. (2017) report that 24 out of the 25 (96%) smalltooth sawfish observed caught from 2008-2015 were released alive. Unlike sea turtles, there are no criteria for assessing the post-release mortality of smalltooth sawfish. However, given the species' biology and the high survival rate of other bottom dwelling shark species (i.e., nurse sharks) caught on bottom longline gear,³⁸ we believe it is very likely that 96% of these animals did survive. Using a mortality rate of 4%, we estimate that of the 6.4 smalltooth sawfish we expect to be caught on an annual basis, there will be 0.3 mortalities (6.4×0.04).

5.2.3.4. Estimated 3-Year Smalltooth Sawfish Takes and Mortalities in Shark Bottom Longline Gear

Every 3 years we estimate (after rounding up to the nearest number) that shark bottom longline gear will catch 20 smalltooth sawfish ($6.4 \times 3 = 19.2$, rounded up). We applied the mortality rates to our 3-year catch estimates. Therefore, of the 20 smalltooth sawfish expected to be caught every 3-years, 0.8 (20×0.04) takes are estimated to result in mortality. Conservatively rounding to the nearest whole number (and because it is not possible to kill a fraction of an animal), we estimate that 1 smalltooth sawfish may be killed every 3 years.

5.2.4 Estimating Smalltooth Sawfish Incidental Catch in Vertical Line Gear

Smalltooth sawfish are occasionally hooked with rod-and-reel during recreational fishing. This catch occurs most frequently in the vicinity of the Everglades National Park and Florida Bay, where the current population is concentrated. North of this area, the number reported caught declines greatly. The National Park Service monitors fishing activity and harvest in Everglades National Park, in part by conducting interviews with anglers and fishing guides at local boat ramps. Most anglers do not report targeting a particular fish species. The target species of the few anglers indicating they do target a particular fish species include snook, spotted sea trout, red drum, and tarpon. All these records are from fishing within state waters, where smalltooth sawfish and sharks are more likely to co-occur.

From 1999-2011, the National Sawfish Encounter Database (NSED) includes 1,399 smalltooth sawfish caught on recreational rod-and-reel gear. Only 15 of those takes occurred in federal waters and none of those 15 caught occurred during trips that reported targeting sharks. Most commonly, no target species were listed ($n=10$), followed by trips targeting snappers and groupers ($n=5$) (NSED unpublished data). The only known smalltooth sawfish catch on rod-and-reel in federal waters while targeting sharks was by an aquaria collector (T. Wiley, pers. comm.). It is unlikely that an incidental catch of a smalltooth sawfish would occur while recreational fishing for any other HMS species besides shark.

³⁸ Of 504 nurse sharks (*Ginglymostoma cirratum*) observed taken from 2007-2009 on bottom longline, 499 (99%) were released alive (Hale et al. 2009, Hale et al. 2010, Hale et al. 2011).

In conducting this consultation, we queried available databases (i.e., the SEFSC Observer Program and Logbook Data, MRIP, NEFSC and GARFO) to see if there were any new records of interactions between HMS vertical line gear and smalltooth sawfish. Although there have been no additional recorded interactions since the information supporting the 2012 Opinion, we concluded there is no reason to change our previous estimates of 1 smalltooth sawfish in recreational vertical line annually, based on the best available data. Both recreational shark fishing effort and smalltooth sawfish abundance are much higher in state waters than in federal waters. We believe it is the reduced effort and smalltooth sawfish abundance in federal waters that make incidental catch of smalltooth sawfish by recreational shark anglers fishing in federal waters rare. Recreational fishing for sharks in the EEZ appears unlikely to catch smalltooth sawfish. However, since 10 of the 15 trips that caught smalltooth sawfish in the EEZ did not indicate a target species, we believe an incidental catch could happen. Even if all 10 of the trips that did not record a target species had been targeting sharks, smalltooth sawfish catch would still be no more than one annually ($10 \text{ sawfish captures} / 13 \text{ years} = 0.8$ rounded up to one sawfish taken per year). Based on this information, we will assume that up to one smalltooth sawfish may be caught annually by recreational fishermen who target sharks in the EEZ, or 3 sawfish every 3 years.

5.2.4.1 Estimating Smalltooth Sawfish Mortality in Vertical Line Gear

Based on the release conditions reported via the NSED, we believe sawfish captured are likely to survive the interactions with HMS vertical line gear. Based on previous interaction observations, we believe all smalltooth sawfish catch by HMS vertical line gear in the future will be released alive with only short-term sublethal effects. Thus we estimate the 3-year incidental take of smalltooth sawfish in HMS vertical line gear is 3, resulting in no mortalities.

5.2.5 Factors Affecting the Likelihood of Exposure of Smalltooth Sawfish to Gillnet Gear

Spatial Overlap of Fishing Effort and Smalltooth Sawfish Abundance

The spatial and temporal overlap of smalltooth sawfish with fishing effort is also a factor that affects the likelihood of these species becoming entangled in shark/smoothhound gillnet gear. The more abundant that animals are in a given area where fishing occurs, the greater the probability that one of them will interact with gear. The temporal distribution of fishing effort and smalltooth sawfish abundance may also be a factor. No smalltooth sawfish were observed caught in Southeast shark gillnet gear from 2004-2016, although a previous catch was observed in 2003.

Net Profile

Both length and profile (i.e., the percentage of the water column spanned by the net) of gillnets in the water column affect the likelihood of smalltooth sawfish exposure to gillnets. Since smalltooth sawfish are predominately a benthic species, they may be more likely to encounter sink gillnets or gillnets set on or near the bottom. Prior to the 2003 observed catch of a smalltooth sawfish in Southeast shark gillnet gear (NMFS 2003a), some people speculated that because these gillnets are set above the seafloor they may not catch

smalltooth sawfish. However, smalltooth sawfish feed on small schooling fish and could occur higher in the water column when engaged in this feeding behavior.

Mesh Size

Smalltooth sawfish may become entangled when their saw penetrates the netting and they try to escape. Smalltooth sawfish can become entangled in any sized mesh, but large mesh is likely particularly problematic. Larger mesh may allow for easier penetration into the gillnetting, thus increasing entanglement potential.

Soak Time

The length of time gillnet gear is left in the water is another important consideration. The longer the soak time, the higher the likelihood smalltooth sawfish may encounter the gillnet gear and become entangled. Since forced submergence is not a concern for smalltooth sawfish, soak times do not appear to impact mortality rates for incidentally caught animals.

Species Morphology

Smalltooth sawfish have unique species morphology that makes them prone to entanglement. See the previous Sections 5.3.1 and 5.3.3 for a description of the morphological features of these species that make them prone to entanglement.

Environmental Conditions

Environmental conditions may also play a large part in whether smalltooth sawfish interact with gillnet gear. Fishing gear can drift according to oceanographic conditions, including wind and waves, surface and subsurface currents, etc.; therefore, depending on these species' behavior, environmental conditions, and location of the set, smalltooth sawfish may become entangled in the gear. There is currently no information on the water temperature preferences of smalltooth sawfish within the action area.

5.2.6 Estimating Smalltooth Sawfish Incidental Catch in Southeast Shark Gillnet Gear

Only one smalltooth sawfish non-lethal take in a shark gillnet has been documented over the 15 years ending in 2017 (Carlson and Richards 2011; NMFS unpublished data). The animal was released in good condition and likely survived the interaction. No smalltooth sawfish catch in shark gillnet gear was observed from 2004-2015 (Carlson and Richards 2011, Carlson et al. 2017, NMFS unpublished data). In conducting this consultation, we queried all available databases for smalltooth sawfish takes by shark and smoothhound gillnet gear in December 2017 (SEFSC logbook, NEFSC observer program, GARFO VTR) in addition to the SEFSC observer program records, to see if there were any new records of interactions between HMS gillnet gear and smalltooth sawfish and we did not find any new records of interactions. While we believe smalltooth sawfish catch in shark gillnet gear is a rare event, the past take leads us to believe another take is possible in the future. However, because this number is so small and because we already rounded up when estimating sawfish takes from hook-and-line gear, which we believe would account for this take, we are not assigning additional take to this component of the fishery. As mentioned previously, gillnet effort targeting LCS and SCS sharks declined as a result of Amendments 2 and 3 to the Consolidated HMS FMP in 2007 and 2010. LCS and SCS

targeted gillnet effort has continued to decline through 2017, such that is extremely limited effort (Carlson and Mathers 2017).

Since the only known shark gillnet take of a smalltooth sawfish was non-lethal, we believe that any take that may also occur in the future, will also be non-lethal.

5.2.7 Summary of Estimated Smalltooth Sawfish Takes and Mortalities from the Proposed Action

In the previous sections we present 3-year estimated takes and mortalities we anticipate under the proposed action based on the analyses we presented in the preceding sections. We chose to present all of the estimates in this manner primarily to help standardize our smalltooth sawfish catch estimates, but also to be consistent with the 3-year approach used in our ITS. By presenting the data in 3-year estimates, we able to consider all of the cumulative takes over time more easily. The numbers of annual catch are likely to fluctuate above and below the number specified from year to year. Thus, we decided to consider all of our take estimates in 3-year periods to incorporate annual variability.

We estimate the total 3-year take in HMS fisheries would be no more than 23 smalltooth sawfish that would result in 1 mortality (Table 5.24).

Table 5.24 Estimated 3-Year Smalltooth Sawfish Takes and Mortalities in HMS Bottom Longline and Vertical Line Gear

Gear	Estimated 3-Year Take	Mortalities
Bottom Longline	20	1
Vertical Line	3	0
Total	23	1

5.3 Effects on Atlantic Sturgeon

Of the gear types used in the HMS fisheries by commercial and/or recreational fishers (i.e., hook-and-line gear, gillnets, purse seines, speargun, harpoon), we believe only HMS gillnet gear may affect and are likely to adversely affect Atlantic sturgeon. This section focuses on evaluating the effects of HMS gillnet gear on Atlantic sturgeon.

5.3.1 Types of Interactions and General Effects from Gillnet Gear

The adverse effects of gillnets on Atlantic sturgeon are likely similar to those experienced by sea turtles and smalltooth sawfish. However, Atlantic sturgeon are morphologically unique. Their cone-shaped snout rapidly transfers meshes over the head and along the body and can cause rapid gilling or wedging. Atlantic sturgeon are also at increased risk of entanglement because their skin is covered in bony scutes. These protrusions increase the likelihood of entanglement and wedging, as the fish attempts to pass through or around gillnets. Larger fish may become wrapped in nets once entangled while they struggle to free themselves. Smaller fish may be entangled by a single monofilament strand hung around a scute (Damon-Randall et al. 2010).

5.3.2 Factors Affecting the Likelihood of Exposure of Atlantic Sturgeon to Gillnet Gear

Spatial Overlap of Fishing Effort and Atlantic Sturgeon Abundance

The spatial and temporal overlap of Atlantic sturgeon with fishing effort is a factor that affects the likelihood of these species becoming entangled in shark gillnet gear. The more abundant that animals are in a given area where fishing occurs, the greater the probability that one of them will interact with gear. The temporal distribution of fishing effort and Atlantic sturgeon abundance may also be a factor.

The best information available on Atlantic sturgeon bycatch in gillnet gear appears to indicate a greater likelihood for interactions during specific times of year. ASMFC (2007) reports that Atlantic sturgeon bycatch across all sink gillnet fisheries was greatest during April and May and lowest during August to October. However, it is important to remember that specific fisheries often operate during certain times of year and in certain regions, so this seasonal bycatch trend could be affected by fishery operations, not necessarily seasonality.

Tie-downs

The use of tie-downs, which create a “pocket” or “bag” effect in gillnets, is also believed to increase the potential for entanglement. Atlantic sturgeon mortality is more likely when tie-downs are in use (ASMFC 2007).

Soak Times

Soak times appear to have a significant impact on the mortality of Atlantic sturgeon. One of the principal findings in the 2006 Sturgeon Technical Committee Workshop on sturgeon bycatch was that soak times greater than 24 hours were associated with substantially higher mortality rates (ASMFC 2007). Numerous scientists have described this relationship, at least in part, finding that increased soak times and increased water temperatures result in higher mortality rates (Collins et al. 1996, Buchanan et al. 2002, Bettoli and Scholten 2006). However, ASMFC (2007) cautions that focusing only on soak time ignores the effect of, or interaction between, other gear variables. This is a concern because some factors like extended soak time and tie-downs are essentially inseparable in observer data (ASMFC 2007).

From 1994-2016, the NEFOP observed 2,267 directed trips for smoothhound. NEFOP observers documented a wide range of soak times during those trips. This observer data indicate the majority of trips (i.e., 96%) have a soak time of less than 24 hours (SEFSC unpublished data).

Mesh Size

Atlantic sturgeon bycatch appears to be relatively closely associated with mesh size. ASMFC (2007) reports 41% of Atlantic sturgeon bycatch was observed in mesh sizes of 5-9.9 inches, 47% of observed Atlantic sturgeon bycatch was in 10-inch mesh or greater; only 12% observed Atlantic sturgeon bycatch was in mesh less than 5 inches. Atlantic sturgeon mortality rates and percent of total mortalities are higher in large mesh fisheries

(7-inches or greater) (36% incidence of mortality)³⁹ but ASMFC (2007) cautions that it is hard to separate the effect of large mesh, tie-downs, and soak time. More specifically, it states tie-downs were used with large mesh nets 74% of the time, and soak times of over 24 hours occurred 79% of the time when tie-downs were used with large mesh. Since both tie-downs and soak time are believed to affect mortality rates in their own right, the extent to which mesh size influence can be attributed to cause of death is limited (ASMFC 2007).

Species Morphology

Atlantic sturgeon have unique species morphology that makes them prone to entanglement. See the introduction to this section for a description of the morphological features of this species that make them prone to entanglement.

Environmental Conditions

Water temperature plays a primary role in triggering the timing of spawning migrations of Atlantic sturgeon (ASMFC 2009). 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); females begin spawning migrations when temperatures are closer to 12°-13°C (54°-55°F) (Dovel and Berggren 1983, Smith 1985, Collins et al. 2000a). These migrations move Atlantic sturgeon out of the marine environment, reducing the potential for entanglement in HMS fisheries. While in the marine environment, Atlantic sturgeon inhabit a wide-range of temperatures. Erickson et al. (2011) reported that for 13 tracked fish, the average monthly water temperatures ranged from 8.3-21.6°C in February and August, respectively. However, two other tracked fish showed much higher (up to 23.9°C) and much lower (down to 5.3°C) temperature ranges (Erickson et al. 2011). This information suggests that the potential for interactions with Atlantic sturgeon in the marine zone exists across a wide range of water temperatures.

5.3.3 Estimating Interactions and Mortality of Atlantic Sturgeon in Smoothhound Gillnet Gear

Summary of Discard Estimates for Atlantic Sturgeon

NEFSC (2011a) explored two approaches to estimate Atlantic sturgeon bycatch in sink gillnet and otter trawl fisheries in the Mid-Atlantic and Northeast. They first evaluated a design-based ratio estimator that used a ratio of total observed sturgeon takes to landings. NEFSC (2011a) concluded this approach would require relying upon a set of assumptions that were too difficult to satisfy given the data available.

NEFSC (2011a) decided to use a generalized linear model to produce a model based estimator instead. A number of models, each evaluating different predictor variables and mesh sizes, were run to identify the model that best fit the available data. NEFSC (2011a) includes models for both otter trawl and sink gillnet gear. However, since the smoothhound fishery only uses sink gillnet gear, we only provide a description of the model's outcome for this gear type. NEFSC (2011a) only used observer data from federal waters, north of Cape Hatteras, North Carolina. Sturgeon included in the analysis included

³⁹ Medium-mesh fisheries (>5- to 7-inch mesh) had a 20% incidence of mortality; small mesh fisheries (≤ 5-inch mesh) had a 2% incidence of mortality (ASMFC 2007).

any animal identified by federal observers as Atlantic sturgeon, as well as any unidentified sturgeon (NEFSC 2011a).

The model-based estimates for sink gillnet gear in NEFSC (2011a) indicated that between 858 and 2,216 Atlantic sturgeon were incidentally caught annually from 2006-2010, with annual mortalities ranging from 30 to 309 animals during the same period. The estimated average mortality rate of Atlantic sturgeon bycaught in sink gillnet gear was 20.6% from 2006-2010 (NEFSC 2011a).

NEFSC (2011a) reports that of the observed Atlantic sturgeon bycatch from 2006-2010, most animals were caught in April and May and the fewest were caught in September and October (NEFSC 2011). These trends are similar to those noted in Stein et al. (2004b) and ASMFC (2007). NEFSC (2011a) also provided an estimate of the total number of Atlantic sturgeon likely caught each year in sink gillnet fisheries, and what proportion of those takes could be attributed to specific federally-managed fisheries. The Northeast Region⁴⁰ Protected Resources Division (NER PRD) and NER Sustainable Fisheries Division discussed the estimates and reallocated some takes based on the knowledge of how certain fisheries operate. However, after the reallocation the estimates of the total number of animals caught annually and the average number of animals caught did not change.⁴¹ Table 5.25 lists the Atlantic sturgeon takes by FMP from 2006-2010, including the reallocated takes. Since smoothhound was not a federally-managed species at the time of analysis, NEFSC (2011a) did not provide an estimate of Atlantic sturgeon takes for that fishery. However, Atlantic sturgeon takes in smoothhound gear are represented as part of the “other” category.

Table 5.25 Estimated Average Atlantic Sturgeon Takes in Sink Gillnet Gear by FMP

Federal FMP	Avg. Annual Gillnet Takes (2006-2010)	Avg. Annual Mortalities (rounded up)* (2006-2010)
Monkfish	719	195
Groundfish	189	39
Bluefish	160	33
Summer Flounder, Scup, & Black Sea Bass	9	2
Spiny Dogfish	107	22
Skate	20	5
Squid, Mackerel, & Butterfish	7	2
Scallop	2	1
All FMPs	1,213	250
Others**	356	74
*Based on the model results gillnet mortalities are assumed to be 20.6%, except in the case of monkfish where the mortality rate is assumed to be 27%		
**“Others” include: smoothhound, croaker, weakfish, striped bass, northern kingfish, and southern kingfish		

⁴⁰ Currently known as the Greater Atlantic Regional Fisheries Office GARFO

⁴¹ In 2016, GARFO received a new set of 5-year discard estimates for Atlantic sturgeon from the NEFSC covering the 2011-2015 time period. That data set is currently being analyzed (William Barnhill, pers. comm.).

5.3.3.1 Observer Data Summary and Estimated Atlantic Sturgeon Catch by Life Stage

We chose to follow the same method in the 2012 Opinion and to use the data from NEFOP on actual interactions between the smoothhound gillnet fishery and Atlantic sturgeon, because it is the best available information on these interactions. Those records indicated that of the Atlantic sturgeon captured by fisheries listed in the NEFSC (2011a)'s "other" category, trips targeting smoothhound sharks with gillnet gear accounted for 30.4% of those takes, or 108 animals.⁴²

It is also important to consider what life stage is being affected and what the impact is to the overall life stage of the species. In general, impacts to adults (i.e., sexually mature animals) are more likely to affect population growth rates than impacts to sub-adults. The NEFSC conducted an analysis of the Atlantic sturgeon takes observed by the NEFOP, categorizing them by length. From 2006-2010, there were 726 observations that could be categorized in this way. Of these, 75% (545) were subadults and 25% (182) were adults; we multiplied our take estimate by these percentages. Using this approach, we estimate that 81 subadults and 27 adults will be captured by gillnet trips targeting smoothhound fishing in federal waters annually.⁴³

5.3.3.2 Estimating Atlantic Sturgeon Mortality in Smoothhound Gillnet Gear

NEFSC (2011a) reports the average Atlantic sturgeon mortality rate in federal sink gillnet fisheries from 2006-2010 was approximately 20.6%. The NEFSC (2011a) report does not provide specific information about Atlantic sturgeon mortality in smoothhound fisheries. We chose to use the mortality estimate from NEFSC (2011a) because it was estimated based on the most recent data available.

In the previous section we estimated the likely number of Atlantic sturgeon that were incidentally captured during smoothhound fishing in federal waters (i.e., 81 subadults and 27 adults annually). To estimate mortality we multiplied those numbers by the 20.6% mortality rate. That calculation indicates that of the estimated 81 subadult takes, 16.7 will be lethal (81×0.206), and of the estimated 27 adult takes, 5.6 will be lethal (27×0.206). Table 5.26 shows the number of likely Atlantic sturgeon takes during smoothhound fishing in federal waters annually, by life stage.

Table 5.26 Annual Adult and Subadult Atlantic Sturgeon (ATS) Takes

Adults			Subadults			Total Takes (Adults and Subadults)
Non-Lethal Takes	Lethal Takes	Total Adult ATS Takes	Non-Lethal Takes	Lethal Takes	Total Subadult ATS Takes	
21.4	5.6	27	64.3	16.7	81	108

⁴² $356 \text{ average annual sink gillnet takes by "other" fisheries} \times 30.4\% \text{ of "other" fishery captures were on smoothhound trips} = 108 \text{ Atlantic sturgeon captures on smoothhound trips.}$

⁴³ $\text{The smoothhound fishery likely takes } 108 \text{ Atlantic sturgeon annually} \times 75\% \text{ likely sub-adults} = 81 \text{ and } 25\% \text{ adults} = 27.$

5.3.3.2.1 Assigning Catch to the Five Atlantic Sturgeon DPSs

Atlantic sturgeon mix extensively in the marine environment, and individuals from all five Atlantic sturgeon DPSs could interact with the smoothhound fishery. Wirgin et al (2015) ran Microsatellite DNA and mitochondrial DNA control-region sequence analyses to determine the population and DPS origin of 173 Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* encountered from the Gulf of Maine to Cape Hatteras, North Carolina, by the NEFOP. Wirgin et al. (2015) found that the Hudson River area had the highest number of specimens as bycatch. Generally, the bycatch represented the geographic province of the river in which they were spawned, but some Atlantic sturgeon, particularly those originating in the South Atlantic DPS, moved great distances (Wirgin et al. 2015).

We used the estimates from Wirgin et al. (2015) because they provide the most recent genetic mixed-stock analysis, which is an accurate approach to determine the DPS and population origin of Atlantic sturgeon bycatch in coastal waters. The mean composition estimates are listed below with the range in parenthesis. These percentages are very similar to the DPS percentages that were used in the 2012 Opinion.

We calculated the number of takes attributable to each DPS based on the following mean % composition estimate for each DPS.

- 2% St. John (Canadian population) (0%-.069%)
- 10% Gulf of Maine DPS (6%-16%)
- 52% New York Bight DPS (43%-59%)
- 12% Chesapeake Bay DPS (7%-18%)
- 2% Carolinas DPS (0-7%)
- 22% South Atlantic DPS (14%-28%)

It important to note that we estimate 0.54 adult and 1.62 sub-adult Atlantic sturgeon takes are likely from the population in St. John, Canada. Since these animals are from a population outside the U.S., which was not listed under the ESA, we do not consider the takes of these animals further in this Opinion. Likewise, since the mean composition estimates do not add to 100, the take estimates in Table 5.27 are slightly less than those estimated above.

Table 5.27 Estimated Annual Atlantic Sturgeon Takes and Mortalities with Smoothhound Gillnet Gear in Federal Waters by DPS

DPS*	Adults			Subadults			Total Takes (Adults and Subadults)
	Non-Lethal Takes	Lethal Takes	Total Takes	Non-Lethal Takes	Lethal Takes	Total Takes	
GOM	2.14	0.56	2.7	6.43	1.67	8.1	10.8
NYB	11.15	2.89	14.04	33.44	8.68	42.12	56.16
CB	2.57	0.67	3.24	7.72	2.0	9.72	12.96
Carolina	0.43	0.11	0.54	1.29	0.33	1.62	2.16
SA	4.72	1.22	5.94	14.15	3.67	17.82	23.76
Total	21.01	5.45	26.46	63.03	16.35	79.38	105.84
GOM = Gulf of Maine DPS, NYB = New York Bight DPS, CB = Chesapeake Bay DPS, and SA = South Atlantic DPS.							
*NOTE: Takes estimated for animals from the St. John, Canada, population are not considered here because they were not listed under the ESA. Because these takes are not considered here, our total take numbers presented in this table are slightly less than the total estimated in Table 5.26.							

Converting Subadults to “Adult Equivalent”

Adult Atlantic sturgeon are generally considered more important to the species than subadults because of their ability to breed. This is an important factor to consider when we evaluate the impacts of the proposed action on Atlantic sturgeon reproduction in our jeopardy analysis (Section 7.0). Thus, we wish to consider not only how the proposed action will affect adults, but also how it would affect subadults that may have lived to become adults (“adult equivalents”). NER-PRD developed an approach for estimating “adult equivalents.” They calculated the proportion of subadults likely to survive to be adults by first adding up the total number of Atlantic sturgeon subadults (i.e., fish ages 2-10) in any year. Then they added up all the adults (i.e., fish ages 11-20). They then divided these sums to get the number of adults per sub-adult. When using the age-variable natural mortality, they estimated that each subadult equates to 0.48 adults. By applying that calculation to our estimates of subadult takes for each DPS from Table 5.33, we can calculate the likely number of adult equivalents that may be captured in smoothhound gear. Since the potential loss of reproduction is an important concern in our jeopardy analysis, and we believe animals suffering non-lethal effects will survive the interaction and could potentially reproduce in the future, we only converted the subadults we anticipate may be lethally taken. Table 5.34 displays the number of adult equivalents for each DPS calculated from the number of lethal subadults takes (rounded).

Table 5.34 Number of Annual DPS Lethal Subadults Takes Converted to Adult Equivalents

DPS	Estimated Lethal Subadult Takes	Subadults Surviving to Adulthood	Estimated Lethal Adult Equivalents Takes
GOM	1.67	0.48	0.80
NYB	8.68	0.48	4.16
CB	2.0	0.48	0.96
Carolina	0.33	0.48	0.16
SA	3.67	0.48	1.76

Discussion of Factors Potentially Influencing the Accuracy of Estimated Past Atlantic Sturgeon Takes in Smoothhound Gillnet Gear

NEFSC (2011a) identified a few assumptions and factors that could influence the accuracy of the past take estimates we used as the basis for our take estimates in the smoothhound fishery. For example, NEFSC (2011a) states the spatial coverage of observed trips is not sufficient to support precise estimation of discards at the level of 3-digit Statistical Area and monthly resolution, but is sufficient to support discard estimation at the level of 2-digit Statistical Areas.⁴⁴ The report also indicated that observer coverage for Mid-Atlantic species was generally lower than coverage rates on Georges Bank and in the Gulf of Maine (NEFSC 2011a). NEFSC (2011a) also considered any observer records where sturgeons were unidentified as Atlantic sturgeon. This means a non-Atlantic sturgeon may have erroneously been counted as one. However, the NEFSC's approach to identifying Atlantic sturgeon is conservative and appropriate since a number of Atlantic sturgeon captures likely go unobserved.

NEFSC (2011a) also states that partitioning incidental takes to FMPs or "others" has limited use because of the high likelihood that incidental takes may be incorrectly attributed to a particular fishery. NEFSC (2011a) reports most trips capture one or more FMP species and the specific gear or deployment patterns within a trip may change. For example, the first half of a trip may have been targeting monkfish and the second half may have been targeting spiny dogfish. If an Atlantic sturgeon was captured on that trip, it may be difficult to tell what the target species was when the incidental take occurred. Additionally, NEFSC (2011a) points out that most of the FMPs manage multiple species. In some cases, the bycatch of Atlantic sturgeon may be more closely associated with one species than the other (e.g., fluke, scup, sea bass) yet they all fall under the same FMP. NER-PRD and NER-Sustainable Fisheries Division discussed the estimates and reallocated some captures based on the knowledge of how certain fisheries operate. The smoothhound gillnet fishery information was included in the catchall category of the fisheries termed "others" in Table 5.25, and the NEFSC estimated the percentage of sturgeon bycatch attributable to smoothhound trips. We believe these estimates are appropriate and conservative for calculating the smoothhound Atlantic sturgeon takes.

5.3.3.2.2 Estimated 3-Year Atlantic Sturgeon Takes and Mortalities in Smoothhound Gillnet Gear by DPS

Conservatively rounding up to the nearest whole number and multiplying by 3 we estimate 34 Gulf of Maine DPS, 170 New York Bight DPS, 40 Chesapeake Bay DPS, 7 Carolina DPS, and 72 South Atlantic DPS will be captured in the smoothhound gillnet gear every 3 years.

Below in Table 5.35, we take the estimates for total takes and lethal takes from Table 5.27 and multiply by 3 and then round that number up to the nearest whole number (because it is not possible to capture a fraction of an animal) to give the estimated number of total and lethal takes of Atlantic sturgeon in smoothhound gillnet gear over a three year period. The

⁴⁴ Data available would not support discard estimation at the level of 3 digit Statistical Areas, (i.e., statistical zone 531, 532, or 626, 627), but allow for discard estimation at the level of 2 digit Statistical Areas (i.e., 53X or 62X).

number of non-lethal takes was calculated by subtracting the number of lethal takes from the total takes.

Table 5.35 Estimated 3-Year Atlantic Sturgeon (ATS) Takes and Mortalities with Smoothhound Gillnet Gear in Federal Waters by DPS

DPS*	Adults			Subadults			Total Takes (Adults and Subadults)
	Non-Lethal Takes	Lethal Takes	Total Takes	Non-Lethal Takes	Lethal Takes	Total Takes	
GOM	7	2	9	19	6	25	34
NYB	34	9	43	100	27	127	170
CB	7	3	10	24	6	30	40
Carolina	1	1	2	4	1	5	7
SA	14	4	18	42	12	54	72
Total	63	19	82	189	52	241	323
GOM = Gulf of Maine DPS, NYB = New York Bight DPS, CB = Chesapeake Bay DPS, and SA = South Atlantic DPS.							

5.3.4 Estimating Interactions and Mortality of Atlantic Sturgeon in Southeast Shark Gillnet Gear

In conducting this consultation, we queried all available databases for Atlantic sturgeon taken in Southeast shark gillnet gear. From 2007-2016, no Atlantic sturgeon incidental catch was observed in Southeast shark gillnet gear. However, in 2002, one Atlantic sturgeon take was observed in Southeast shark gillnet gear; the animal was released alive. In 2011, four Atlantic sturgeon were documented by observers from the SGOP. Those incidental catches occurred during sets targeting finfish, not sharks. Two of these animals were released alive, two were dead. This information indicates that Atlantic sturgeon catch in shark directed gillnet sets are uncommon but they do occur and have occurred recently in similar gears. There is very limited available information on the potential future impacts of shark gillnet gear on Atlantic sturgeon. For example, estimating the average number of Atlantic sturgeon takes from the number of observed takes in shark gillnet gear from 2002-2016 indicates less than 0.1 animals would be taken annually. Conversely, the information on observed takes in other gillnet fisheries indicates Atlantic sturgeon takes could be as high as four a year. Because of this uncertainty, we acted conservatively and estimated that two Atlantic sturgeon may be taken annually by Southeast shark gillnet gear, the estimate in the 2012 Opinion.

Since approximately 2005, no Southeast shark gillnet effort has occurred north of Virginia. The vast majority of the shark gillnet fishing effort occurs in what is defined as Marine Mixing Zone 3 and the southern part of Marine Mixing Zone 2. NER-PRD (2012) determined that Atlantic sturgeon from the New York Bight and South Atlantic DPSs

comprised the greatest proportion of animals in Marine Mixing Zone 2.⁴⁵ NER-PRD (2012) concluded that there was not enough information available to estimate the proportions of animals in Marine Mixing Zone 3. Therefore, NER-PRD (2012) recommends using the information from the observer program in Marine Mixing Zone 2 to describe the likely proportion in Marine Mixing Zone 3. Fish from the Carolina DPS appeared very rarely, if ever, in observer datasets. However, we believe that since most shark gillnet fishing occurs in the waters adjacent to the Carolina DPS, it is reasonable to conclude that an Atlantic sturgeon from this DPS could be captured during Southeast shark gillnet fishing. Therefore, we estimate that of the two Atlantic sturgeon takes that may occur each year in shark gillnet gear, one is likely to be an individual from the Carolina DPS. We believed the other is likely from the South Atlantic DPS because most of the species observed in Marine Mixing Zone 2 are either from the New York Bight (49%) and South Atlantic (20%) DPSs. Although a greater proportion of individuals from the New York Bight have been observed across the whole of Marine Mixing Zone 2, we believe it more likely that the takes will be of individuals from the South Atlantic DPS because the New York Bight is relatively far north of the area fished while the SA DPS borders the Carolina DPS.

The available information indicates shark gillnet mortality rates of Atlantic sturgeon range from 0 to 50%. However, NEFSC 2011 indicates that mortality rates for sink gillnet (which are sometimes used in this fishery) are much lower, between 20 and 27%. ASMFC (2007) reports mortality rates for sink gillnet gear with mesh sizes of 5 inches or greater to be 20-36%. Since incidental take mortality rates may be as high 50%, we conservatively estimate that one of the predicted annual takes may be lethal. Although most of the fishing is done off the area occupied by the Carolina DPS, we expect the exposure to this gear between the Carolina DPS and the South Atlantic DPS to be similar. The Carolina DPS has fewer fish than the South Atlantic DPS, which could mean the South Atlantic DPS would have more exposure, however, the fishing will be more concentrated in areas where the Carolina DPS is found. We have no reason to believe an animal from the Carolina DPS would be more prone to mortality in a shark gillnet than an animal from the South Atlantic DPS, meaning we have no way of determining whether the lethal take is likely to affect the South Atlantic DPS or the Carolina DPS. Therefore, we will act conservatively and assume each take is lethal.

The estimated ratio of subadults to adults is 3 to 1. This would suggest that, all things being equal, these takes are three times more likely to be subadults than adults. However, we have chosen to act conservatively and will assume that these lethal takes will be adults. Thus, we anticipate the Southeast shark gillnet fisheries may take one adult Atlantic sturgeon from the South Atlantic DPS and one from the Carolina DPS each year.

⁴⁵ The NER-PRD MSA (NER-PRD 2012) estimated the following DPS composition for Atlantic sturgeon in Marine Mixing Zone 2: 2% St. John (Canadian population); 11% Gulf of Maine DPS, 49% New York Bight DPS, 14% Chesapeake Bay DPS, and 20% South Atlantic DPS.

5.3.4 Summary of Estimated Atlantic Sturgeon DPS Takes and Mortalities from the Proposed Action

In Table 5.36, we present 3-year estimated takes and mortalities we anticipate under the proposed action based on the analyses we presented in the preceding sections. We chose to present all of the estimates in this manner primarily to help standardize our Atlantic Sturgeon DPS catch estimates, but also to be consistent with the 3-year approach used in our ITS. By presenting the data in 3-year estimates, we able to consider all of the cumulative takes over time more easily. The numbers of annual catch are likely to fluctuate above and below the number specified from year to year. Thus, we decided to consider all of our take estimates in 3-year periods to incorporate annual variability.

We estimate the total take in the HMS fisheries would be no more than 328 Atlantic sturgeon, comprised of 34 Gulf of Maine DPS, 169 New York Bight DPS, 40 Chesapeake Bay DPS, 10 Carolina DPS, and 75 South Atlantic DPS caught every 3 years. Applying our overall mortality rates and conservatively rounding up the final numbers, we estimate that up to 76 Atlantic sturgeon, comprised of 8 Gulf of Maine DPS, 36 New York Bight DPS, 8 Chesapeake Bay DPS, 5 Carolina DPS, and 19 South Atlantic DPS may be killed, every 3 years.

Table 5.36 Estimated 3-Year Atlantic Sturgeon DPS Total Take (T) and Mortality (M) Estimates of Adults and Subadults in the HMS Fisheries

DPS*	Adults Caught in Smoothhound Gillnet Gear		Subadults Caught in Smoothhound Gillnet Gear		Adults Caught in Southeast Shark Gillnet Gear		Total Takes and Mortalities (Adults and Subadults)	
	T	M	T	M	T	M	T	M
GOM	9	2	25	6	0	0	34	8
NYB	43	9	127	27	0	0	170	36
CB	10	3	30	6	0	0	40	9
Carolina	2	1	5	1	3	3	10	5
SA	18	4	54	12	3	3	75	19
Total	82	19	241	52	6	6	329	77
GOM = Gulf of Maine DPS, NYB = New York Bight DPS, CB = Chesapeake Bay DPS, and SA = South Atlantic DPS.								

5.4 Effects on the Central and Southwest Atlantic DPS of Scalloped Hammerhead Shark

Of the gear types used in the HMS fisheries by commercial and/or recreational fishers (i.e., hook-and-line gear, gillnets, harpoon, purse seines, and speargun), we believe certain vertical hook-and-line gear (rod and reel, bandit rigs, buoy gear, and handlines) may affect and are likely to adversely affect scalloped hammerhead sharks. This section focuses on evaluating the effects of those vertical hook-and-line gear on the Central and Southwest Atlantic DPS of scalloped hammerhead shark.

5.4.1 Types of Interactions and General Effects from Hook-and-Line Gear

Hook-and-line gear fishing affects scalloped hammerhead sharks primarily by hooking, but also by entanglement and trailing of gear. Hooking and entanglement can lead to cuts, puncture wounds, mouth or other tissue damage, and animals can suffer from the stress of the capture. Hooked or entangled sharks may potentially also suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns, though we have no actual evidence of such effects.

5.4.2 Factors Affecting the Likelihood of Exposure of Scalloped Hammerhead Sharks to Hook-and-Line Gear

Gear usage and fishing techniques (soak times)

The amount of fishing effort affects the landing of scalloped hammerheads that are accidentally caught by HMS fisheries. Number of fishers, number of trips, and length of time gear is left in the water (soak times) are all important considerations. More fishing increases the probability of hooking this species.

Spatial overlap of fishing effort and scalloped hammerhead sharks

The location of the fishery in relation to the species is a factor influencing the likelihood that the HMS fisheries will interact with and hook a scalloped hammerhead. The range of the Central and Southwest Atlantic DPS of scalloped hammerhead in U.S. waters relevant to the proposed action is exclusively in Caribbean waters. Only that portion of the fishery that occurs in the federal waters of the species' range is subject to effects from the fishery's gear. Data indicate that there is hook-and-line gear used within the range of the Central and Southwest Atlantic DPS of scalloped hammerhead shark.

5.4.3 Estimating Interactions with Scalloped Hammerhead Sharks in Vertical Line Gear

As described in Chapter 2, commercial HMS fisheries are limited in the Caribbean. Most of the fishing in the Caribbean is recreational or artisanal vertical line. Because of the limited geographic area of the Caribbean, the small-scale of commercial HMS fisheries, the small size of the vessels involved, the relatively low number of known participants, and the use of traditional handgear, HMS fisheries in the Caribbean are considered to have negligible impacts to ocean and coastal habitats (NMFS 2012).

The majority of small-scale commercial vessels participating in HMS fisheries in the Caribbean Region use handgear (handline, rod and reel). The limited possession of fishing permits and dealer permits and reporting of recreational catch has resulted in limited catch and landings data from the U.S. Caribbean fisheries. However, some of these fishermen have federal permits for other species (i.e., snapper, grouper, pelagics) and are required to report all landings, including shark, due to the regulations of these fisheries. Trip-ticket data from Puerto Rico and the USVI offers the best source of shark landings data, specifically in the U.S. Caribbean fisheries, where sharks are rarely targeted, but rather caught as bycatch. NOAA's SEFSC is currently working on estimating the Caribbean commercial and recreational data sets (trip ticket data) from Puerto Rico and USVI. Most of the fishing around Puerto Rico occurs in territorial waters, which extend out to 9 nm.

About 95.3% of the fishable area in the U.S. Caribbean is in territorial waters and about 4.7% of the fishable area is in the U.S. Caribbean EEZ (NMFS 2012). Landings data is available from the HMS Caribbean Small Boat Permit collected by NMFS in cooperation with territorial government fisheries data collection programs. No scalloped hammerhead shark landings have ever been reported in the HMS Caribbean Small Boat Permit landings data.

MRIP data from Puerto Rico from 2001 through 2016 show an expanded catch of 688 scalloped hammerhead sharks landed with vertical line gear by recreational charter boats within the territorial waters (10 miles/8.7 nautical miles) of Puerto Rico. This total only includes catch with no target species recorded. MRIP is not conducted in other parts of the U.S. Caribbean so all available data is from Puerto Rico. In addition, following the 2017 hurricane season, MRIP has been placed on hold in Puerto Rico. All scalloped hammerhead sharks were caught in territorial waters. The scalloped hammerhead lengths were all <1000mm, in the 600-900mm range. Expanded estimates are 516 scalloped hammerhead sharks caught in 2003, 44 caught in 2004, 30 caught in 2006, and 98 caught in 2012, with proportional standard errors (PSEs) ranging from 79.9 to 100.⁴⁶ This results in an annual average of 43 sharks taken for the years with data (from 2001 to 2016, 688 shark/16 years = 43 shark/year).

We attribute this expanded scalloped hammerhead shark catch to those fishing for HMS species (because no target fishery was listed). We then assume that those fishing for HMS species use the entire fishable area in territorial and federal waters in the U.S. Caribbean and catch a proportional amount of sharks in those waters. Given these assumptions, we can estimate scalloped hammerhead sharks takes coming from HMS targeted trips in federal waters. Federal waters make up 4.7% of the fishable area in the U.S. Caribbean, and territorial waters the remaining 95.3%. Therefore, we assume 2.1 scalloped hammerhead sharks were caught annually by HMS fisheries in the federal waters (43 sharks caught in territorial waters = 95.3 % of total sharks caught by HMS fishers in territorial and federal waters; total = $43/0.953 = 45.1$; $45.1 \text{ total sharks} * 4.7\% \text{ caught in federal waters} = 2.1 \text{ sharks caught in federal waters}$).

Although there were no recorded captures from federal waters, we believe this is an appropriate estimate of scalloped hammerhead shark catch in Caribbean federal waters. The fishable area is considered to be within the 100 fathom contour (NMFS 2012), which as stated 4.7% of that area is federal waters. Waters past the 100 fathom contour tend to be less productive and can be harder to access due to the small size of the majority of vessels. However, as these vessels can easily access federal waters to the 100 fathom contour from the territorial waters, and many of these vessels troll, it is highly likely that many vessels venture into federal waters while on a trip. We also believe this is a conservative estimate because we used the expanded estimate of scalloped hammerhead sharks caught in Puerto Rico territorial waters by all recreational fishing vessels (charter boat) using vertical line reported when no target fishery was listed. It is unlikely that all of these trips were targeting HMS species, and thus, this could overestimate interactions. However, given the

⁴⁶ PSE, or proportional standard error, expresses the standard error of an estimate as a percentage of the estimate and is a measure of precision. A PSE value greater than 50 indicates a very imprecise estimate.

lack of data from places other than Puerto Rico and the lack of observer coverage in the Caribbean in general, we have not assumed take from waters off the USVI as we have no information on such takes. We are using this as the best available estimate for recreational vertical line in federal waters, and are not assuming any additional take from commercial gear in federal waters, given the lack of data on interactions and the nature of the fishery.

5.4.3.1 Estimating Scalloped Hammerhead Shark Mortality in Vertical Line Gear

Based on the above, we estimate there will be 2.1 interactions annually between scalloped hammerhead shark and certain vertical line gear in the federal HMS fisheries. In this section, we estimate the annual number of mortalities resulting from these interactions.

Scalloped hammerhead sharks are biologically vulnerable to interactions with fishing gear. These sharks are obligate ram ventilators and suffer very high at-vessel fishing mortality in bottom longline fisheries (Morgan and Burgess 2007, Macbeth et al. 2009). From 1994-2005, NMFS observers calculated that out of 455 scalloped hammerheads caught on commercial bottom longline vessels in the northwest Atlantic and Gulf of Mexico, 91.4% were dead when brought aboard. Size did not seem to be a factor influencing susceptibility as 70% of the young *S. lewini* (0-65 cm), 95.2% of the juveniles (66-137cm), and 90.9% of the adults (>137cm) suffered at-vessel fishing mortality. Soak time of the longline had a positive effect on the likelihood of death (Morgan and Burgess 2007), with soak times longer than 4 hours resulting in > 65% mortality (Morgan et al. 2009). When soak time was shortened to 1 hour, *S. lewini* at-vessel fishing mortality decreased to 12% (Lotti 2011). Lotti (2011) also found that at-vessel fishing mortality was negatively correlated with *S. lewini* length ($p=0.0032$) and dissolved oxygen ($p=0.003$), with male scalloped hammerheads showing a higher probability of suffering from at-vessel mortality compared to females ($p=0.0265$).

We do not have mortality estimates for hammerhead sharks in vertical line gear. Rod and reel fishing is more akin to pelagic longline fishing than the bottom longline fishing discussed above as the rod and reel fishers tend to fish in the water column and not on the bottom (e.g. trolling). A review of Atlantic pelagic longline gear observer data from 1992-2015 shows that 90 out of 169 scalloped hammerhead sharks caught were dead (SEFSC unpublished data). This indicates a mortality rate of 0.53. Because the vertical line gears in the Caribbean are likely to have much shorter soak times (less than an hour) than longline gear (hours), assuming the same mortality rate in the vertical longline gear could over-estimate mortalities, though we are not certain without additional information. However, we believe this is a reasonable approach to estimating lethal take. In the absence of observer data or more specific information on mortalities from this gear type, this is the best available information upon which to estimate mortalities. We estimate that of the 2.1 scalloped hammerhead sharks caught annually, 1.1 could be killed (2.1×0.53).

5.4.4 Summary of Estimated Central and Southwest Atlantic DPS of Scalloped Hammerhead Shark Takes and Mortalities from the Proposed Action

In the previous section, we concluded that HMS vertical line gear could take 2.1 scalloped hammerhead sharks resulting in 1.1 mortalities annually. Thus, every 3 years we expect

6.3 scalloped hammerheads will be caught by the fishery. Of these takes, 3.3 are expected to result in mortalities. Because it is not possible to take a fraction of a shark, we estimate the total 3-year take in the HMS fisheries would be no more than 7 Central and Southwest Atlantic DPS of scalloped hammerhead sharks that would result in 4 mortalities.

5.5 Effects on Oceanic Whitetip Sharks

Of the gear types used in the HMS fisheries by commercial and/or recreational fishers (i.e., hook-and-line gear, gillnets, harpoon, speargun, and purse seines), we believe certain vertical hook-and-line gear (rod and reel, bandit, buoy gear, and handline) may affect and are likely to adversely affect oceanic whitetip sharks. This section focuses on evaluating the effects of those vertical hook-and-line gears on oceanic whitetip sharks.

5.5.1 Types of Interactions and General Effects from Hook-and-Line Gear

Certain vertical hook-and-line gears (rod and reel, bandit, buoy, and handline gears) are likely to adversely affect oceanic whitetip sharks. Hook-and-line gear fishing affects oceanic whitetip sharks primarily by hooking, but also by entanglement and trailing of gear. Hooking and entanglement can lead to cuts, puncture wounds, mouth or other tissue damage, and animals can suffer from the stress of the capture. Hooked or entangled sharks may potentially also suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns, though we have no actual evidence of such effects.

5.5.2 Factors Affecting the Likelihood of Oceanic Whitetip Shark Exposure to Hook-and-Line Gear

Gear usage and fishing techniques (soak times)

The amount of fishing effort affects the likelihood of oceanic whitetip sharks being incidentally caught on the HMS vertical hook-and-line gear noted above. Number of fishers, number of trips, and length of time the gear is left in the water (soak time) are all important considerations. More fishing increases the probability of hooking this species.

Spatial overlap of fishing effort and oceanic whitetip sharks

The location of the fishing effort in relation to the species is a factor influencing the likelihood that HMS fisheries will interact with and hook an oceanic whitetip shark. The oceanic whitetip shark ranges throughout the Atlantic Ocean and the Gulf of Mexico. Only effort that occurs in deep open ocean areas in federal waters has the potential to affect oceanic whitetip shark.

5.5.3 Estimating Interactions with Oceanic Whitetip Sharks in Vertical Line Gear

5.5.3.1 Estimating Oceanic Whitetip Shark Interactions in Commercial Vertical Hook-and-Line Gear

The only commercial catch records of oceanic whitetip sharks are in HMS buoy gear data. Between 2006 and 2015, there were 13 oceanic whitetips reported caught in buoy gear targeting swordfish (SEFSC unpublished data). Over this time period there was a total of

88,723 hooks deployed with HMS buoy gear in the Atlantic and Gulf of Mexico, with an annual average of 8,872 hooks. The total number of oceanic whitetip sharks per 1,000 hooks was 0.15 ($13/88,723 \times 1,000$). The average effort of the fishery has been variable over the timeframe analyzed. The average from the five years ending 2017 of effort data available is 9,883 hooks deployed annually. Therefore, if we assume the fishery will fish at a similar rate in the future, we estimate that 1.5 oceanic whitetip sharks will be taken by HMS buoy gear annually ($9,883 \times 0.15/1000$). We do not have records of catch of oceanic whitetip sharks in commercial HMS rod and reel, bandit, and handline gear. Although we believe interactions with these gears are possible, in the absence of data on those interactions, we cannot estimate take. We also do not believe it is appropriate to estimate take with these gears from interactions with buoy gear. Unlike rod and reel, which is normally trolled in the fisheries that have the potential to interact with oceanic whitetip shark, and bandit gear and handlines, which are used in target fishing, mostly in schools of tuna, buoy gear floats baits for swordfish which are not schooling fish.

5.5.3.2 Estimating Oceanic Whitetip Shark Interactions in Recreational Vertical Hook-and-Line Gear

From 2001-2016, the only records of oceanic whitetip sharks in the MRIP recreational fisheries data from a trip that may have been targeting HMS species was from Puerto Rico territorial waters in 2015. MRIP is not conducted in other parts of the U.S. Caribbean so all available data is from Puerto Rico. In addition, following the 2017 hurricane season, MRIP has been placed on hold in Puerto Rico. The SEFSC expanded estimate of this catch is 67 oceanic whitetip sharks caught in 2015, the only catch in the 2001 to 2016 time series, resulting in an average of 4.2 oceanic whitetips sharks caught annually in the Caribbean ($67 \text{ sharks}/16 \text{ years} = 4.2 \text{ sharks per year}$). The PSE for this estimate is 105.5, so this is also a very imprecise estimate. As mentioned in Section 5.4.3, approximately 4.7% of the fishable area in the U.S. Caribbean is in the EEZ. Based on the same assumptions described in Section 5.4.3, we can assume 0.2 oceanic whitetips sharks were caught by HMS fisheries annually in the Caribbean EEZ ($4.2 \text{ sharks in territorial waters} = 0.953(x)$; $x \text{ (total sharks in territorial and federal waters)} = 4.4$; $0.047(4.4) = 0.2$).

5.5.3.3 Estimated Oceanic Whitetip Shark Mortality in Vertical Line Gear

Based on the information above, we estimate 1.7 oceanic whitetip shark ($1.5 + 0.2$) interactions annually from buoy gear and recreational hook-and-line gear associated with the proposed action. In this section we determine the annual number of estimated mortalities resulting from these interactions. The susceptibility of sharks in general to immediate or post-release mortality varies by species (Gallagher et al. 2014) and gear type (de Silva et al. 2001; Francis et al. 2001; Moyes et al. 2006).

We do not have mortality estimates for oceanic whitetip sharks in vertical line gear. Buoy gear and rod and reel fishing is akin to pelagic longline fishing as these gears tend to fish in the water column. Based on observer data from 1992-2000, of the 131 oceanic whitetip sharks observed in pelagic longline fishery of southeastern U.S., 95 were alive and 36 were dead at haul-back, which equates to an at-vessel mortality rate of 27.5% (Beerkircher et al. 2002). Another study from the U.S. pelagic longline fishery estimates at-vessel mortality

of ~22.7% (Gallagher et al. 2015). Similarly, a study from pelagic longline fisheries in the South Atlantic (but not from the U.S. pelagic longline fishery) showed a range of 11-28% at-vessel mortality rates for oceanic whitetip shark (Fernandez-Carvalho et al. 2015). A recent review of Atlantic pelagic longline gear observer data from 1992–2015 shows that 89 out of 207 oceanic whitetip sharks caught were dead, which indicates a mortality rate of 0.43 (SEFSC unpublished data). These rates are relatively low compared to other species (e.g., scalloped hammerhead) but do not account for post-release mortality, which remains uncertain. Therefore, we believe a mortality rate of 43% is an appropriate estimate to use for this analysis and is a conservative estimate compared to the previous studies. Based on the mortality rate of 43%, we estimate of the 1.7 oceanic whitetip sharks caught annually on vertical line gear that 0.7 could be killed (1.7×0.43).

5.5.4 Summary of Estimated Oceanic Whitetip Shark Takes and Mortalities from the Proposed Action

In the previous section, we concluded that HMS vertical line gear could take 1.7 oceanic whitetip sharks annually, resulting in 0.7 mortalities annually. Thus, every 3 years we expect 5.1 ($1.7 \times 3 = 5.1$) oceanic whitetip sharks will be caught by the fishery. Of these takes, 2.1 (0.7×3) are expected to result in mortalities. Because it is not possible to take a fraction of a shark, rounding this number results in 3 oceanic whitetip sharks killed every 3 years. Therefore, we estimate the total 3-year take in the HMS fisheries associated with the proposed action would be no more than 6 oceanic whitetip sharks that would result in 3 mortalities.

5.6 Effects on Giant Manta Ray

Of the gear types used in the HMS fisheries by commercial and/or recreational fishers (i.e., hook-and-line gear, gillnets, harpoon, purse seines, and spearguns), we believe certain hook-and-line gear (namely, bottom longline gear) and gillnet gear may affect and are likely to adversely affect giant manta rays. This section focuses on evaluating the effects of bottom longline gear and gillnets on giant manta rays.

5.6.1 Types of Interactions and General Effects from Hook-and-Line Gear

We believe certain hook-and-line gear, namely bottom longline gear, is likely to adversely affect giant manta rays. Hook-and-line gear fishing affects giant manta rays primarily by hooking, but also by entanglement and trailing of gear. Hooking and entanglement can lead to cuts, puncture wounds, mouth or other tissue damage, and animals can suffer from the stress of the capture. Hooked or entangled manta rays may potentially also suffer impaired swimming or foraging abilities.

5.6.2 Factors Affecting the Likelihood of Giant Manta Ray Exposure to Hook-and-Line Gear

Gear usage and fishing techniques (soak times)

The amount of fishing effort may affect the likelihood of giant manta rays being accidentally caught on HMS hook-and-line gear. Number of fishers, number of trips, and

length of time the gear is left in the water (soak time) are all important considerations. More fishing increases the probability of hooking this species.

Spatial overlap of fishing effort and giant manta rays

The location of fishing effort in relation to the species is a factor influencing the likelihood that the HMS hook-and-line gear will interact with and hook a giant manta ray. The giant manta ray ranges throughout the Atlantic Ocean and the Gulf of Mexico. The giant manta ray can be found in shallow nearshore waters as well as deep offshore waters.

5.6.3 Estimating Interactions with Giant Manta Rays in Bottom Longline Gear

5.6.3.1 Estimating Interactions with Giant Manta Rays in Bottom Longline Gear

Giant manta rays have been recorded as catch in shark bottom longline gear, in the research fishery. Giant manta rays have not been reported as catch from other federal HMS bottom longline fisheries. Between 2008 and 2016, there were 2 giant manta rays reported caught in the shark bottom longline research fishery (SEFSC unpublished data). As mentioned previously, this fishery has 100 percent observer coverage. Therefore, based on our assumption that the fishery will fish at a similar rate in the future, we estimate that 0.2 giant manta rays will be taken by HMS bottom longline gear annually ($2/9 = 0.2$).

5.6.3.2 Estimated Giant Manta Ray Mortality in Bottom Longline Gear

Based on the information above, we estimated 0.2 giant manta ray will be captured on bottom longline gear annually. In this section, we will estimate the number of mortalities annually. There is very limited information on immediate or post-release mortality of mobula species. The two giant manta rays caught on shark bottom longline gear in the research fishery were released alive. A recent review of bottom longline (Atlantic shark and Gulf reef fish) and gillnet (Atlantic shark and Atlantic coastal migratory pelagic) observer data from 2008–2016 shows that a total of 3 giant manta rays were caught and released alive (SEFSC unpublished data). These rates do not account for post-release mortality, which remains uncertain. Based on this information, we believe there are no giant manta ray mortalities from HMS bottom longline gear.

5.6.4 Factors Affecting the Likelihood of Giant Manta Ray Exposure to Shark Gillnet Gear, Including Smoothhound Gillnet Gear

Spatial Overlap of Fishing Effort and Giant Manta Ray Abundance

The spatial and temporal overlap of giant manta rays with fishing effort is a factor that affects the likelihood of these species becoming entangled in shark/smoothhound gillnet gear. The more abundant that animals are in a given area where fishing occurs, the greater the probability that one of them will interact with gear. The temporal distribution of fishing effort and giant manta ray abundance may also be a factor.

Species Morphology

The conditions faced by manta rays during the different phases of capture in fishing operations include traumatic handling practices (lifting up by the gills or dragging on the deck and/or towing). Giant manta rays may also be exposed to physical contact with hard

objects, the harsh harvesting process of removing it from the fishing gear and removal from the water (lack of oxygen, exposure to the sun and organs crushed because of the weight of gravity). Manta rays are large and thus it can be extremely difficult to lift them back into the water.

Environmental Conditions

Water temperature may play a role in the timing of giant manta ray migrations and presence at aggregation sites. More research is needed to understand the movements of giant manta rays and potential interactions with gillnets during various times of the year. Initial studies seem to show a seasonal component to their movements.

5.6.5 Estimating Interactions with Giant Manta Rays in Shark Gillnet Gear, Including Smoothhound Gillnet Gear

Of the commercial records for gillnet gears, giant manta rays have been documented as catch only in shark gillnet gear (there are no known interactions of giant manta rays in smoothhound gear). The SEFSC estimated the level of protected resource take from 2008-2016 in shark gillnet gear (strike, drift, anchor, and other) that were either targeting Atlantic sharks or smoothhound sharks (Carlson and Mathers 2017). In the following sections, we describe the NMFS unpublished observer data and the take estimate calculated in that report. Carlson and Mathers 2017 include more detailed discussion of the data sources used, calculation methods, constraints of those methods, and the assumptions under which those calculations were made.

5.6.5.1 Observer Data Summary

Between 2008 and 2016, one giant manta ray was observed caught in shark gillnet gear, including smoothhound gillnet gear, in 2012 (NMFS unpublished data).

5.6.5.2 Extrapolated Giant Manta Ray Catch in Shark Gillnet Gear

Carlson and Mathers (2017) expanded the take estimate of giant manta rays caught in the shark fishery in the Gulf of Mexico and South Atlantic from 2008-2016 to 23.9 (based on the one individual caught in 2012). Based on this estimate, the annual take estimate is 2.7 (23.9/9).

5.6.5.3 Estimated Giant Manta Ray Mortality in Shark Gillnet Gear

All of the giant manta rays caught in federally managed gillnet gear from 2008-2016 were released alive (NMFS unpublished data). Unlike sea turtles, there are no criteria for assessing the post-release mortality of giant manta rays. However, given the species' biology and the likelihood that the species would survive after being released from a gillnet, we believe it is very likely that all of these animals did survive; therefore, we estimate zero giant manta ray mortalities.

5.6.6 Summary of Estimated Giant Manta Rays Takes and Mortalities from the Proposed Action

In the previous section, we concluded that HMS bottom longline gear could take 0.2 giant manta rays annually, resulting in 0 mortalities annually. We also concluded that HMS gillnet gear could take 2.7 giant manta rays annually, resulting in 0 mortalities annually.

Thus, every 3 years we expect 9 ($0.2+2.7*3=8.7$) giant manta rays will be caught by the fisheries associated with the proposed action. None of these takes are expected to result in mortalities. Therefore, we estimate the total 3-year take in HMS fisheries associated with the proposed action would be no more than 9 giant manta rays that would result in 0 mortalities.

6.0 Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area of 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.

Human-induced mortality and/or injury of sea turtles, smalltooth sawfish, Atlantic sturgeon, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays occurring in the action area are reasonably certain to occur in the future. The sources of those effects include vessel interactions, marine debris, pollution, global climate change, and coastal development. While the combination of these activities may prevent or slow the recovery of populations of sea turtles, smalltooth sawfish, Atlantic sturgeon, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays, the magnitude of these effects is currently unknown.

6.1 Vessel Interactions

NMFS's STSSN data indicate that vessel interactions are responsible for a large number of sea turtles stranding within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded sea turtles have obvious propeller or collision marks (Dwyer et al. 2003). Still, it is not always clear whether the collision occurred pre- or post-mortem. We believe that sea turtle injuries and mortalities by vessel interactions will continue in the future.

NMFS has received anecdotal reports that giant manta rays may be affected by vessel interactions in aggregation areas on the East Coast of Florida evidenced by scarring. There is no evidence that these vessel interactions have caused any giant manta ray mortalities in the action area.

Because smalltooth sawfish are benthic species and Atlantic sturgeon, scalloped hammerhead sharks, and oceanic whitetip sharks are distributed throughout the water column, vessel strikes are not considered a threat to them in the action area.

6.2 Pollution

Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them on sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. However, the level of impacts cannot be projected. Marine debris (e.g., discarded fishing line or lines from boats) can entangle sea turtles in the water and drown them. Sea turtles commonly ingest plastic or mistake debris for food. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging behavior. As mentioned previously, sea turtles are not very easily affected by changes in water quality or increased

suspended sediments, but if these alterations make habitat less suitable for sea turtles and hinder their capability to forage, eventually they would tend to leave or avoid these areas (Ruben and Morreale 1999).

Noise pollution has been raised primarily as a concern for marine mammals (including ESA-listed large whales) but may be a concern for other marine organisms, including sea turtles and ESA-listed fish. The potential effects of noise pollution on sea turtles, smalltooth sawfish, and Atlantic sturgeon range from minor behavioral disturbance to injury and death. The noise level in the ocean is thought to be increasing at a substantial rate due to increases in shipping and other activities, including seismic exploration, offshore drilling, and sonar used by military and research vessels. Concerns about noise in the action area of this consultation include increasing noise due to increasing commercial shipping and recreational vessels.

6.3 Global Climate Change

Global climate change is likely adversely affecting sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events and fluctuation of precipitation levels, and change in air and water temperatures. The effects on ESA-listed species are unknown at this time. There are multiple hypothesized effects to ESA-listed species, including changes in their range and distribution as well as prey distribution and/or abundance due to water temperature changes. Ocean acidification may also negatively affect marine life, particularly organisms with calcium carbonate shells that serve as important prey items for many species. Global climate change may also affect reproductive behavior in sea turtles, including earlier onset of nesting, shorter intervals between nesting, and a decrease in the length of nesting season. Sea level rise may also reduce the amount of nesting beach available. Changes in air temperature may also affect the sex ratio of sea turtle hatchlings. A decline in reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of sea turtles in the Atlantic.

Sea levels and water temperatures are expected to rise, and levels of precipitation are likely to fluctuate. Drought and inter- and intra-state water allocations and their associated impacts to Atlantic sturgeon will continue and may intensify. A rise in sea level may drive the salt wedge upriver on river systems inhabited by Atlantic sturgeon, potentially constricting Atlantic sturgeon habitat. NMFS will continue to work with states to implement ESA Section 6 agreements, and with researchers holding Section 10 permits, to enhance programs to quantify and mitigate these takes and effects.

6.5 Coastal Development

Within the action area, beachfront development, lighting, and beach erosion potentially reduce or degrade sea turtle nesting habitats or interfere with hatchlings movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. Coastal counties are presently adopting stringent protective measures to

protect hatchling sea turtles from the disorienting effects of beach lighting. Some of these measures were drafted in response to lawsuits brought against the counties by concerned citizens who charged the counties with failing to uphold the ESA by allowing unregulated beach lighting that results in takes of hatchlings.

Beyond the threats noted above, NMFS is not aware of any proposed or anticipated changes in other human-related actions (e.g., poaching, habitat degradation) or natural conditions (e.g., overabundance of land or sea predators, changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on the sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays covered by this Opinion.

7.0 Jeopardy Analyses

The analyses conducted in the previous sections of this Opinion serve to provide a basis to determine whether the proposed action would be likely to jeopardize the continued existence of sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, or giant manta rays. In Section 5, we outlined how the proposed action would affect these species at the individual level and the extent of those effects in terms of the number of associated interactions, captures, and mortalities of each species to the extent possible with the best available data. Now we assess each of these species' response to this impact, in terms of overall population effects, and whether those effects of the proposed action, in the context of the status of the species (Section 3), the environmental baseline (Section 4), and the cumulative effects (Section 6), are likely to jeopardize their continued existence in the wild.

To "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 the recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Thus, in making this determination for each species, we must look at whether the proposed action directly or indirectly reduces the reproduction, numbers, or distribution of a listed species. Then if there is a reduction in 1 or more of these elements, we evaluate whether it would be expected to cause an appreciable reduction in the likelihood of both the survival and the recovery of the species.

The NMFS and USFWS's ESA Section 7 Handbook (USFWS and NMFS 1998) defines survival and recovery, as they apply to the ESA's jeopardy standard. Survival means "the species' persistence... beyond the conditions leading to its endangerment, with sufficient resilience to allow recovery from endangerment." Survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a sufficiently large 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. The Section 7 Handbook defines recovery as "improvement in the status of a 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." Recovery is the process by which species' ecosystems are restored and/or threats to the species are removed so self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

The status of each listed species or DPS likely to be adversely affected by the proposed action is reviewed in Section 3. For any species listed globally, our jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery at the global species range. For any species listed as DPSs, a jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery of that DPS.

7.1 Sea Turtles

Some sea turtle species are listed as a single species distributed globally; therefore, a jeopardy determination must find the proposed action will appreciably reduce the likelihood of such species' survival and recovery at the scale of its global range. Nine DPSs for loggerheads and 11 DPSs for green sea turtles have been identified. The loggerhead DPS likely to be adversely affected by the proposed action is the Northwest Atlantic DPS, listed as threatened. Two green sea turtle DPSs (North Atlantic DPS and South Atlantic DPS) may occur in the action area, and are likely to be adversely affected by the proposed action. Therefore, for loggerhead and green sea turtles, a jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery of these DPSs in the wild.

7.1.1 Loggerhead Sea Turtles (NWA DPS)

The proposed action may result in up to 91 loggerhead sea turtle takes, 40 of which are expected to be nonlethal and 51 of which are expected to be lethal, every 3 years. The potential nonlethal capture and release of 40 loggerhead sea turtles every 3 years is not expected to have a measurable impact on the reproduction, numbers, or distribution of this species. The individuals suffering nonlethal injuries are expected to fully recover such that no reductions in reproduction or numbers of loggerhead sea turtles are anticipated. The takes may occur anywhere in the action area, and the action area encompasses a very small portion of the overall range/distribution of the NWA DPS of loggerhead sea turtles. Since any incidentally caught animal would be released within the general area where caught, no change in the distribution of loggerhead sea turtles is anticipated.

The estimated maximum of 51 lethal takes every 3 years associated with the proposed action represents a reduction in numbers. These lethal takes would also result in a future reduction in reproduction as a result of lost reproductive potential, as some of these individuals would be females that 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 3 or 4 clutches of eggs every 2-4 years, with 100-130 eggs per clutch. Thus, the loss of adult female sea turtles 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 takes attributed to the proposed action. Because all the potential interactions are expected to occur at random throughout the proposed action area, which accounts for a very small fraction of the species' overall range, the distribution of loggerhead sea turtles is expected to be unaffected.

Whether the reductions in loggerhead sea turtle numbers and reproduction attributed to the proposed action would appreciably reduce the likelihood of survival for loggerheads depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends, i.e., whether the estimated reductions, when viewed within the context of the environmental baseline, status of the species, and cumulative effects are of such an extent that adverse effects on population dynamics are appreciable. In Section 3.3.2, we reviewed the status of the species in terms of nesting and female

population trends and several of the most recent assessments based on population modeling (i.e., (Conant et al. 2009; NMFS-SEFSC 2009)). Below, we synthesize what that information means in general terms and in the more specific context of the proposed action.

Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerhead sea turtles require high survival rates throughout their life to maintain a population. In other words, late-maturing species cannot tolerate much anthropogenic mortality without going into decline. Conant et al. (2009) concluded that loggerhead natural growth rates are small, natural survival needs to be high, and even low to moderate mortality can drive the population into decline. Because recruitment to the adult population takes many years, population modeling studies suggest even small increased mortality rates in adults and subadults could substantially impact population numbers and viability (Chaloupka and Musick 1997; Crouse et al. 1987; Crowder et al. 1994).

SEFSC (2009) estimated the minimum adult female population size for the NW Atlantic DPS in the 2004-2008 timeframe to likely be between approximately 20,000-40,000 individuals (median 30,050), with a low likelihood of being as many as 70,000 individuals. Another estimate for the entire western North Atlantic population was a mean of 38,334 adult females using data from 2001-2010 (Richards et al. 2011). A much less robust estimate for total benthic females in the western North Atlantic was also obtained, with a likely range of approximately 30,000-300,000 individuals, up to less than 1 million.

SEFSC (2011) preliminarily estimated the loggerhead population in the Northwestern Atlantic Ocean along the continental shelf of the Eastern Seaboard during the summer of 2010 at 588,439 individuals (estimate ranged from 381,941 to 817,023) based on positively identified individuals. The NMFS-NEFSC's point estimate increased to approximately 801,000 individuals when including data on unidentified sea turtles that were likely loggerheads. The NMFS-NEFSC (2011) underestimates the total population of loggerheads since it did not include Florida's east coast south of Cape Canaveral or the Gulf of Mexico, which are areas where large numbers of loggerheads can also be found. In other words, it provides an estimate of a subset of the entire population.

Florida accounts for more than 90% of U.S. loggerhead nesting. The Florida Fish and Wildlife Conservation Commission conducted a detailed analysis of Florida's long-term loggerhead nesting data (1989-2015). They indicated that following a 24% increase in nesting between 1989 and 1998, nest counts declined sharply from 1999 to 2007. However, annual nest counts showed a strong increase (74%) from 2008 to 2015. Examining only the period between the high-count nesting season in 1998 and 2017, researchers found a slight but nonsignificant increase, indicating a reversal of the post-1998 decline. The overall change in counts from 1989 to 2015 was significantly positive (38%); however, it should be noted that wide confidence intervals are associated with this complex data set (<http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/>).

Abundance estimates accounting for only a subset of the entire loggerhead sea turtle population in the western North Atlantic indicate the population is large (i.e., several hundred thousand individuals). Nesting trends have been significantly increasing over several years. Additionally, our estimate of future takes is not a new source of impacts on the species. The same or a similar level of captures has occurred in the past, yet we have still seen positive trends in the status of this species.

The proposed action could remove up to 51 individuals every 3 years. These removed individuals represent approximately 0.013% every 3 years of the low end of the NMFS-SEFSC (2011) estimate of 381,941. As we noted above, this estimate reflects a subset of the entire loggerhead population in the western North Atlantic Ocean, and thus these individuals may represent an even smaller proportion of the population removed. While the loss of 51 individuals every 3 years is an impact to the population, in the context of the overall population's size and current trend, it would not be expected to result in a detectable change to the population numbers or trend. The amount of loss is likely smaller than the error associated with estimating (through extrapolation) the overall population in the 2011 report. Consequently, we expect the western North Atlantic population to remain large (i.e., hundreds of thousands of individuals) and to retain the potential for recovery, and the proposed action to not cause the population to lose genetic heterogeneity, broad demographic representation, or successful reproduction, nor affect loggerheads' ability to meet their lifecycle requirements, including reproduction, sustenance, and shelter. Thus, we conclude the proposed action is not likely to appreciably reduce the likelihood of this DPS's survival in the wild.

The loggerhead recovery plan defines the recovery goal as "...ensure[ing] that each recovery unit meets its Recovery Criteria alleviating threats to the species so that protection under the ESA is no longer necessary" (NMFS and USFWS 2008). The plan then identifies 13 recovery objectives needed to achieve that goal. Elements of the proposed action support or implement the specific actions needed to achieve a number of these recovery objectives. Thus, we do not believe the proposed action impedes the progress of the recovery program or achieving the overall recovery strategy.

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2009) was written prior to the loggerhead sea turtle DPS listings. However, this plan deals with the populations that comprise the current NWA DPS and is therefore, the best information on recovery criteria and goals for the DPS. The plan lists the following recovery objectives that are relevant to the effects of the proposed action:

Objective No. 1: Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females

Objective No 2: Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes

Objective No. 10: Minimize bycatch in domestic and international commercial and artisanal fisheries

Objective No 11: Minimize trophic changes from fishery harvest and habitat alteration

The recovery plan anticipates that, with implementation of the plan, the western North Atlantic population will recover within 50-150 years, but notes that reaching recovery in only 50 years would require a rapid reversal of the then-declining trends of the NRU, PFRU, and NGMRU. The minimum end of the range assumes a rapid reversal of the current declining trends; the higher end assumes that additional time will be needed for recovery actions to bring about population growth.

Recovery Objective No. 1, “Ensure that the number of nests in each recovery unit is increasing...,” is the plan’s overarching objective and has associated demographic criteria. Nesting trends in most recovery units have been significantly increasing over several years. As noted previously, we believe the future takes predicted will be similar to the levels of take that have occurred in the past and those past takes did not impede the positive trends we are currently seeing in nesting during that time. We also indicated that the potential lethal take of 51 loggerhead sea turtles over the future every 3 years is so small in relation to the overall population, that it would be hardly detectable. For these reasons, we do not believe the proposed action will impede achieving this recovery objective. Continuation of the proposed action is not counter to the recovery plan’s Objective No.s 2 and 10: “ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes” and “minimize bycatch in domestic and international commercial and artisanal fisheries.” While bycatch may still occur during the proposed action, we have no indication that it is affecting the in-water abundance of juveniles related to strandings, and bycatch minimization measures are in place in these fisheries that avoid or minimize lethal bycatch. For these reasons, we do not believe the proposed action will impede achieving these recovery objectives.

Continuation of the proposed action is also not counter to Objective No 11: “minimize trophic changes from fishery harvest and habitat alteration.” There is no indication the HMS fisheries analyzed in this opinion are causing any trophic changes that would affect loggerhead sea turtles. For these reasons, we do not believe the proposed action will impede achieving this recovery objective.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NWA DPS of the loggerhead sea turtle in the wild.

7.1.2 Kemp’s Ridley Sea Turtles

The proposed action may result in up to 22 Kemp’s ridley sea turtle takes, of which 11 are expected to be lethal and 11 are expected to be nonlethal, every 3 years. The nonlethal capture of 11 Kemp’s ridley sea turtles every 3 years is not expected to have any

measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of Kemp's ridley sea turtles are anticipated. The takes may occur anywhere in the action area and the action area encompasses a tiny portion of Kemp's ridley sea turtles' overall range/distribution. Since any incidentally caught animals would be released within the general area where caught, no change in the distribution of Kemp's ridley sea turtles is anticipated.

The lethal take of up to 11 Kemp's ridley sea turtles every 3 years would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. The TEWG (1998a) estimates age at maturity from 7-15 years. Females return to their nesting beach about every 2 years (TEWG 1998a). The mean clutch size for Kemp's ridleys is 100 eggs/nest, with an average of 2.5 nests/female/season. Lethal captures 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. While we have no reason to believe the proposed action will disproportionately affect females, the loss of up to 11 Kemp's ridley sea turtles every 3 years, could preclude the production of thousands of eggs and hatchlings, of which a fractional percentage is expected to survive to sexual maturity. Thus, the death of any females would eliminate their contribution to future generations, and result in a reduction in sea turtle reproduction. The anticipated captures are expected to occur anywhere in the action area and sea turtles generally have large ranges; thus, no reduction in the distribution of Kemp's ridley sea turtles is expected from the take of these individuals.

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. In the section on the Status of Species, we presented the status of the Kemp's ridley sea turtle, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the section on the Environmental Baseline, we considered the past and present impacts of all state, federal, or private actions and other human activities in, or having effects in, the action areas that have affected and continue to affect this DPS. In the section on Cumulative Effects, we considered the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action areas.

In the absence of any total population estimates for Kemp's ridley sea turtles, nesting trends are the best proxy we have for estimating population changes. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley sea turtle nests in Mexico reached a record high of 21,797 in 2012 (Gladys Porter Zoo nesting database 2013). In 2013 through 2014, there was a second significant decline in Mexico nests, with only 16,385 and 11,279 nests recorded, respectively. In 2015, nesting in Mexico improved to 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Gladys Porter Zoo 2016). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm. to NMFS SERO PRD, August 31, 2017) and a slight

decline in nests in 2018. A small nesting population is also emerging in the U.S., primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (NPS data, <http://www.nps.gov/pais/naturescience/strp.htm>, <http://www.nps.gov/pais/naturescience/current-season.htm>). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015-2017.

It is important to remember that with significant inter-annual variation in nesting data, sea turtle population trends necessarily are measured over decades and the long-term trend line better reflects the population increase in Kemp's ridleys. With the recent increase in nesting data (2015-17) and recent declining numbers of nests (2010; 2013-14), it is too early to tell whether the long-term trend line is affected. Nonetheless, data from 1990 to present continue to support that Kemp's ridley sea turtles are showing a generally increasing nesting trend. We believe this long-term increasing trend in nesting is evidence of an increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Additionally, our evaluation of potential future mortalities is based on our belief that the same level of interactions occurred in the past, and even with that level we have still seen positive trends in the status of this species. After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the species discussed in this Opinion, we believe the potential loss of up to 11 Kemp's ridley sea turtles every 3 years will not have any detectable effect on the population, distribution or reproduction of Kemp's ridley sea turtles. Therefore, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

The Kemp's ridley recovery plan defines the recovery goal as: "...conserve[ing] and protect[ing] the Kemp's ridley sea turtle so that protections under the Endangered Species Act are no longer necessary and the species can be removed from the List of Endangered and Threatened Wildlife" (NMFS et al. 2011b). The recovery plan for the Kemp's ridley sea turtle (NMFS et al. 2011c) lists the following relevant recovery objective:

Objective: 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.

With respect to this recovery objective, the nesting numbers in 2018, indicate there were a total of 17,945 nests on the main nesting beaches in Mexico. This number represents approximately 7,178 nesting females for the season based on 2.5 clutches/female/season. The number of nests reported annually from 2010 to 2014 overall declined; however, they rebounded in 2015 through 2017, and declined again in 2018. Although there has been a substantial increase in the Kemp's ridley population within the last few decades, the number of nesting females is still below the number of 10,000 nesting females per season

required for downlisting (NMFS and USFWS 2015). Since we concluded that the potential loss of up to 11 Kemp's ridley sea turtles every 3 years is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede the progress toward achieving this recovery objective. Nonlethal captures of these sea turtles would not affect the adult female nesting population or number of nests per nesting season. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of Kemp's ridley sea turtles' recovery in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the Kemp's ridley sea turtle in the wild.

7.1.3 Green Sea Turtles (North Atlantic and South Atlantic DPS)

Mixed-stock analyses of foraging grounds show that green sea turtles from multiple nesting beaches commonly mix at feeding areas across the Caribbean and Gulf of Mexico, with higher contributions from nearby large nesting sites and some contribution estimated from nesting populations outside the DPS (Bass et al. 1998; Bass and Witzell 2000; Bjorndal and Bolten 2008; Bolker et al. 2007). In other words, the proportion of animals on the foraging grounds from a given nesting beach is proportional to the overall importance of that nesting beach to the entire DPS. For example, Tortuguero, Costa Rica, is the largest nesting beach in the North Atlantic DPS and the number of animals from that nesting beach on foraging grounds were higher than from any other nesting beach. More specifically, Lahanas et al. (1998) showed that juvenile green sea turtles in the Bahamas originate mainly from the western Caribbean (Tortuguero, Costa Rica) (79.5%) (North Atlantic DPS) but that a significant proportion may be coming from the eastern Caribbean (Aves Island/Suriname; 12.9%) (South Atlantic DPS).

Flipper tagging studies provide additional information on the co-mingling of turtles from the North Atlantic DPS and South Atlantic DPS. Flipper tagging studies on foraging grounds and/or nesting beaches have been conducted in Bermuda (Meylan et al. 2011), Costa Rica (Troeng et al. 2005), Cuba (Moncada et al. 2006), Florida (Johnson and Ehrhart 1996; Kubis et al. 2009), Mexico (Zurita et al. 2003; Zurita et al. 1994), Panama (Meylan et al. 2011), Puerto Rico (Collazo et al. 1992; Patricio et al. 2011), and Texas (Shaver 1994; Shaver 2002). Nesters have been satellite tracked from Florida, Cuba, Cayman Islands, Mexico, and Costa Rica. Troeng et al. (2005) report that while there is some crossover of adult female nesters from North Atlantic DPS into the South Atlantic DPS, particularly in the equatorial region where the DPS boundaries are in closer proximity to each other, North Atlantic DPS nesters primarily use the foraging grounds within the North Atlantic DPS.

As discussed in section 3.3.5, within U.S. waters, individuals from both the North Atlantic and South Atlantic DPSs can be found on foraging grounds. While there are currently no in-depth studies available to determine the percent of North Atlantic and South Atlantic DPS individuals in any given location, an analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of Mexico) found approximately 4% of individuals

came from nesting stocks in the South Atlantic DPS. On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the South Atlantic DPS (Bass and Witzell 2000). All of the individuals in both studies were benthic juveniles.

Taken together, this information suggests that the vast majority of the anticipated captures in the Gulf of Mexico and South Atlantic regions are likely to come from the North Atlantic DPS. However, it is possible that animals from the South Atlantic DPS could be captured during the proposed action. For these reasons, we will act conservatively and conduct 2 jeopardy analyses, 1 for each DPS. In Section 5.1.5 we estimated catch of up to 46 green sea turtles from the North Atlantic DPS, of which 25 are expected to be lethal and 21 are expected to be nonlethal and an estimated catch of up to 3 green sea turtles from the South Atlantic DPS, of which 2 are expected to be lethal and 1 is expected to be nonlethal, every 3 years.

7.1.2.1 North Atlantic DPS

The proposed action may result in 46 green sea turtle takes from the North Atlantic DPS (21 nonlethal, 25 lethal) every 3 years. The potential nonlethal capture of 21 green sea turtles from the North Atlantic DPS every 3 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. The individuals suffering nonlethal injuries are expected to fully recover such that no reductions in reproduction or numbers of green sea turtles are anticipated. The takes may occur anywhere in the action area, which encompasses only a tiny portion of green sea turtles' overall range/distribution within the North Atlantic DPS. Because any incidentally caught animal would be released within the general area where caught, no change in the distribution of North Atlantic DPS green sea turtles is anticipated.

The potential lethal take of 25 green sea turtles from the North Atlantic DPS every 3 years would reduce the number of North Atlantic green sea turtle DPS, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. Lethal takes would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have survived otherwise to reproduce. For example, an adult green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every 2-4 years, with 110-115 eggs/nest, of which a small percentage is expected to survive to sexual maturity. The anticipated lethal takes 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 green sea turtles within the North Atlantic DPS is expected from these captures.

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. In Section 3.3.5, we presented and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. Below we review the details of that information.

Seminoff et al. (2015) estimated that there are greater than 167,000 nesting females in the North Atlantic DPS. The nesting at Tortuguero, Costa Rica, accounts for approximately 79% of that estimate (approximately 131,000 nesters), with Quintana Roo, Mexico, (approximately 18,250 nesters; 11%), and Florida, USA, (approximately 8,400 nesters; 5%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015).

At Tortuguero, Costa Rica, the number of nests laid per year from 1999 to 2003, was approximately 104,411 nests/year, which corresponds to approximately 17,402-37,290 nesting females each year (Troëng and Rankin 2005). That number increased to an estimated 180,310 nests during 2010; corresponding to 30,052-64,396 nesters. This increase has occurred despite substantial human impacts to the population at the nesting beach and at foraging areas (Campell and Lagueux 2005; Troëng 1998; Troëng and Rankin 2005).

Nesting locations in Mexico along the Yucatan Peninsula also indicate the number of nests laid each year has increased (Seminoff et al. 2015). In the early 1980s, approximately 875 nests/year were deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS 2007a). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpublished data, 2013, in Seminoff et al. 2015)

In Florida, most nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). As described in Section 3.3.5, nesting has increased substantially over the last 20 years and peaked in 2017 with 38,954 nests statewide. In-water studies conducted over 24 years in the Indian River Lagoon, Florida, suggest similar increasing trends, with green sea turtle captures up 661% (Ehrhart et al. 2007). Similar in-water work at the St. Lucie Power Plant site revealed a significant increase in the annual rate of capture of immature green sea turtles over 26 years (Witherington et al. 2006).

In summary, nesting at the primary nesting beaches has been increasing over the course of decades. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for North Atlantic DPS green sea turtles is clearly increasing, we believe the potential lethal take of 25 North Atlantic DPS green sea turtles every 3 years attributed to the proposed action will not have any measurable effect on that trend. Therefore, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the North Atlantic DPS of green sea turtle in the wild.

The North Atlantic DPS of green sea turtles does not have a separate recovery plan at this time. However, an Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) does exist. Since the animals within the North Atlantic DPS all occur in the Atlantic Ocean and would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide

until a new plan, specific to the North Atlantic DPS, is developed. The Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.

Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

According to data collected from Florida's index nesting beach survey from 1989-2015, green sea turtle nest counts across Florida have increased approximately ten-fold from a low of 267 in the early 1990s to a high of 27,975 in 2015 (<http://myfwc.com/research/wildlife/sea-turtles/nesting/2015-nesting-trends/>). There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting, however, it is likely that numbers on foraging grounds have also increased.

The potential lethal take of up to 25 North Atlantic DPS green sea turtles every 3 years will result in a reduction in numbers when captures occur, but it is unlikely to have any detectable influence on the recovery objectives and trends noted above. Nonlethal captures of these sea turtles would not affect the adult female nesting population or number of nests per nesting season. Thus, the proposed action will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of North Atlantic DPS green sea turtles' recovery in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the North Atlantic DPS of the green sea turtle in the wild.

7.1.2.3 South Atlantic DPS

The proposed action may result in up to 3 green sea turtle captures from the South Atlantic DPS (1 nonlethal, 2 lethal) every 3 years. The potential nonlethal capture of 1 South Atlantic DPS green sea turtle every 3 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. The individuals suffering nonlethal injuries are expected to fully recover such that no reductions in reproduction or numbers of green sea turtles are anticipated. The takes may occur anywhere in the action area and the action area encompasses a tiny portion of green sea turtles' overall range/distribution within the South Atlantic DPS. Since any incidentally caught animal would be released within the general area where caught, no change in the distribution of South Atlantic DPS green sea turtles is anticipated.

The potential lethal take of 2 green sea turtles every 3 years would reduce the number of green sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. Lethal interactions would also result in a

potential reduction in future reproduction, assuming the individuals caught would at least in some years be female and would have survived otherwise to reproduce. For example, an adult green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every 2-4 years, with 110-115 eggs/nest, of which a small percentage is expected to survive to sexual maturity. 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 green sea turtles within the South Atlantic DPS is expected from these captures.

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. In Section 3.3.5, we summarized available information on number of nesters and nesting trends at South Atlantic DPS beaches. Seminoff et al. (2015) estimated that there are greater than 63,000 nesting females in the South Atlantic DPS, though they noted the adult female nesting abundance from 37 beaches could not be quantified. The nesting at Poilão, Guinea-Bissau, accounted for approximately 46% of that estimate (approximately 30,000 nesters), with Ascension Island, United Kingdom, (approximately 13,400 nesters; 21%), and the Galibi Reserve, Suriname (approximately 9,400 nesters; 15%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015).

Seminoff et al. (2015) reported that while trends cannot be estimated for many nesting populations due to the lack of data, they could discuss possible trends at some of the primary nesting sites. Seminoff et al. (2015) indicated that the nesting concentration at Ascension Island (United Kingdom) is one of the largest in the South Atlantic DPS and the population has increased substantially over the last 3 decades (Broderick et al. 2006; Glen et al. 2006). At Ascension Island Mortimer and Carr (1987) counted 5,257 nests in 1977 (about 1,500 females), and 10,764 nests in 1978 (about 3,000 females) whereas from 1999–2004, a total of about 3,500 females nested each year (Broderick et al. 2006). Since 1977, numbers of nests on 1 of the 2 major nesting beaches, Long Beach, have increased exponentially from around 1,000 to almost 10,000 (Seminoff et al. 2015). From 2010 to 2012, an average of 23,000 nests per year was laid on Ascension (Seminoff et al. 2015). Seminoff et al. (2015), caution that while these data are suggestive of an increase, historic data from additional years are needed to fully substantiate this possibility.

Seminoff et al. (2015) reported that the nesting concentration at Galibi Reserve and Matapica in Suriname was stable from the 1970s through the 1980s. From 1975–1979, 1,657 females were counted (Schulz 1982), a number that increased to a mean of 1,740 females from 1983–1987 (Ogren 1989b), and to 1,803 females in 1995 (Weijerman et al. 1998). Since 2000, there appears to be a rapid increase in nest numbers (Seminoff et al. 2015).

In the Bijagos Archipelago (Poilão, Guinea-Bissau), Parris and Agardy (1993 as cited in Fretey, 2001) reported approximately 2,000 nesting females per season from 1990 to 1992, and Catry et al. (2002) reported approximately 2,500 females nesting during the 2000 season. Given the typical large annual variability in green sea turtle nesting, Catry et al.

(2009) suggested it was premature to consider there to be a positive trend in Poilão nesting, though others have made such a conclusion (Broderick et al. 2006). Despite the seeming increase in nesting, interviews along the coastal areas of Guinea-Bissau generally resulted in the view that sea turtles overall have decreased noticeably in numbers over the past two decades (Catty et al. 2009). In 2011, a record estimated 50,000 green sea turtle clutches were laid throughout the Bijagos Archipelago (Seminoff et al. 2015).

Nesting at the primary nesting beaches has been increasing over the course of the decades. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for green sea turtles is clearly increasing, we believe the potential lethal take of 2 South Atlantic DPS of green sea turtles every 3 years attributed to the proposed action will not have any measurable effect on that trend. Therefore, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the South Atlantic DPS of green sea turtle in the wild.

Like the North Atlantic DPS, the South Atlantic DPS of green sea turtles does not have a separate recovery plan in place at this time. However, an Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) does exist. Since the animals within the South Atlantic DPS all occur in the Atlantic Ocean and would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide until a new plan, specific to the South Atlantic DPS, is developed. In our analysis for the North Atlantic DPS, we stated that the Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.

Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

The nesting recovery objective is specific to the North Atlantic DPS, but demonstrates the importance of increases in nesting to recovery. As previously stated, nesting at the primary South Atlantic DPS nesting beaches has been increasing over the course of the decades. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting; however, it is likely that numbers on foraging grounds have also increased.

The potential lethal take of up to 2 South Atlantic DPS green sea turtles every 3 years will result in a reduction in numbers when captures occur, but it is unlikely to have any detectable influence on the trends noted above. Nonlethal captures of sea turtles would not affect the adult female nesting population or number of nests per nesting season. Thus, the proposed action is not in opposition to the recovery objectives above and will not result in an appreciable reduction in the likelihood of the South Atlantic DPS of green sea turtles' recovery in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the South Atlantic DPS of the green sea turtle in the wild.

7.1.4 Leatherback Sea Turtles

The proposed action may result in up to 7 leatherback sea turtle takes, 4 of which are expected to be lethal, every 3 years. The nonlethal capture of 3 leatherback sea turtles every 3 years, is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of leatherback sea turtles is anticipated.

The lethal take of up to 4 leatherback sea turtles every 3 years would reduce the population by that number compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Lethal captures could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and would have survived otherwise 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. While we have no reason to believe the proposed action will disproportionately affect females, the death of any female leatherbacks that would have survived otherwise to reproduce would eliminate its and its future offspring's contribution to future generations. The anticipated lethal interactions are expected to occur anywhere in the action area. Given these sea turtles generally have large ranges, no reduction in the distribution of leatherback sea turtles is expected from the proposed action.

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. In the section on the Status of Species, we presented the status of the leatherback sea turtle, outlined threats, and discussed information on nesting. In the section on the Environmental Baseline, we considered the past and present impacts of all state, federal, or private actions and other human activities in, or having effects in, the action areas that have affected and continue to affect this species. In the section on Cumulative Effects, we considered the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action areas.

The Leatherback Turtle Expert Working Group estimated 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 based on 2004 and 2005 nesting count data (TEWG 2007). The potential loss of up to 3 leatherback sea turtles every 3 years accounts for only 0.00005-0.0001% of the

North Atlantic population estimates, which is a subset of the listed entity. We do not believe these potential losses will have any detectable impact on these population numbers.

Of the 15 leatherback nesting populations in the North Atlantic, 7 show an increase in nesting (Florida, Puerto Rico [not Culebra], St. Croix-U.S. Virgin Islands, British Virgin Islands, Trinidad, Guyana, and Brazil) and 3 have shown a decline in nesting (Puerto Rico [Culebra], Costa Rica [Tortuguero], and Costa Rica [Gandoca]) from 2009 to 2015. The most important nesting populations (French Guiana and Suriname) have remained stable. Suriname and French Guiana may represent over 40% of the world's leatherback nesting population (Spotila et al. 1996), accounting for between 31,000 to 60,000 nests annually (NMFS and USFWS 2013).

The main nesting areas in Puerto Rico 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 9 nests recorded in 1978 and to a minimum of 469-882 nests recorded each year between 2000 and 2005 (NMFS and USFWS 2013). However since 2004, nesting has steadily declined in Culebra, which appears to reflect a shift in nest site fidelity rather than a decline in the female population (NMFS and USFWS 2013).

In the U.S. Virgin Islands, St. Croix (Sandy Point National Wildlife Refuge), leatherback nesting was estimated to increase at 13% per year from 1994 through 2001. However, nesting data from 2001 through 2010 indicate nesting has slowed, possibly due to fewer new recruits and lowered reproductive output (NMFS and USFWS 2013). The average annual growth rate was calculated as approximately 1.1 (with an estimated confidence interval between 1.07 and 1.13) using the number of observed females at Sandy Point, St. Croix, from 1986 to 2004 (TEWG 2007).

In Costa Rica, Tortuguero, leatherback nesting has decreased 88.5% overall from 1995 through 2011 (NMFS and USFWS 2013). Troëng et al. (2007) estimated a 67.8% overall decline from 1995 through 2006. However, these estimates are based on an extrapolation of track survey data, which has consistently underestimated the number of nests reported during the surveys (NMFS and USFWS 2013). Regardless of the method used to derive the estimate, the number of nests observed over the last 17 years has declined. Troeng et al. (2004) found a slight decline in the number of nests at Gandoca, Costa Rica, between 1995 and 2003, but the confidence intervals were large. Data between 1990 and 2004 at Gandoca averaged 582.9 (+ 303.3) nests each year, indicating nest numbers have been lower since 2000 (Chacón-Chaverri and Eckert 2007), and the numbers are not increasing (TEWG 2007).

Aside from the nesting declines in Tortuguero, which are significant, most of the other nesting populations appear to be increasing or are remaining stable, including the most significant populations in French Guiana and Suriname. Since we anticipate a low number of mortalities every 3 years and we have no reason to believe nesting females will be disproportionately affected, we believe the potential mortalities associated with the proposed action will have no detectable effect on current nesting trends.

Since we do not anticipate the proposed action will have any detectable impact on the population overall, or current nesting trends, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

The Atlantic recovery plan for the U.S. population of the leatherback sea turtles (NMFS and USFWS 1992) lists the following relevant recovery objective:

Objective: 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. As noted previously, the Florida and St. Croix nesting populations are increasing. The nesting population in Culebra, Puerto Rico, had been increasing since the late 1970s but has been declining in recent years; however, it appears these declines may reflect a shift in nest site fidelity rather than a decline in the female population. Since we concluded that the potential loss of up to 4 leatherback sea turtles every 3 years is not likely to have any detectable effect on these nesting trends, we do not believe the proposed action is impeding the progress toward achieving this recovery objective. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the leatherback sea turtle in the wild.

7.1.5 Hawksbill Sea Turtles

The proposed action may result in up to 2 hawksbill sea turtle takes, with 1 expected to be lethal, every 3 years. The nonlethal capture of 1 hawksbill sea turtle every 3 years, is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of hawksbill sea turtles is anticipated.

The lethal take of up to 1 hawksbill sea turtles every 3 years would reduce the number of hawksbill sea turtles, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Any potential lethal interaction could also result in a reduction in future reproduction, assuming the individual would be a female and would have survived to reproduce in the future. For example, an adult hawksbill sea turtle can lay 3-5 clutches of eggs every few years (Meylan and Donnelly 1999; Richardson et al. 1999) with up to 250 eggs/nest (Hirth and Latif 1980). Thus, the loss of a female could preclude the production of thousands of eggs

and hatchlings, of which a fraction would otherwise survive to sexual maturity and contribute to future generations. The anticipated lethal interactions are expected to occur anywhere in the action area. Given these sea turtles generally have large ranges, no reduction in the distribution of hawksbill sea turtles is expected from the proposed action.

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. In the section on the Status of Species, we presented the status of the hawksbill sea turtle, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends. In the section on the Environmental Baseline, we considered the past and present impacts of all state, federal, or private actions and other human activities in, or having effects in, the action areas that have affected and continue to affect this species. In the section on Cumulative Effects, we considered the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action areas.

In the absence of any total population estimates for hawksbill sea turtles, nesting trends are the best proxy we have for estimating population changes. The most recent 5-year status review estimated between 22,000 and 29,000 adult females existed in the Atlantic basin in 2007 (NMFS 2013b); this estimate does not include juveniles of either sex or mature males. The potential loss of up to 1 hawksbills every 3 years would equal only 0.0045% of the adult female population, which is only a portion of the entire population. Hawksbill nesting trends also indicate an improvement over the last 20 years. A survey of historical nesting trends (i.e., 20-100 years ago) for the 33 nesting sites in the Atlantic Basin found declines at 25 of those sites and data were not available for the remaining 8 sites. However, in the last 20 years, nesting trends have been improving. Of those 33 sites, 10 sites now show an increase in nesting, 10 sites showed a decrease, and data for the remaining 13 are not available (NMFS 2013b).

Our evaluation of the impact of future captures is based in part on our belief that the same level of capture occurred in the past, yet we have still seen positive trends in the status of this species. We believe increases in nesting over the last 20 years, relative to the historical trends, indicate improving population numbers. Additionally, even when we conservatively evaluate the potential effects of the proposed action on a portion of the hawksbill population (i.e., adult females) we believe the impacts will be minor relative to the entire population. Thus, we believe the potential loss of up to 1 hawksbill sea turtles every 3 years will not have any detectable effect on the population, distribution or reproduction of hawksbills. Therefore, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

The Recovery Plan for the population of the hawksbill sea turtles (NMFS and USFWS 1993) lists the following relevant recovery objectives over a period of 25 continuous years:

Objective: The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests on at least 5 index

beaches, including Mona Island (Puerto Rico) and Buck Island Reef National Monument (U.S. Virgin Islands).

Objective: The numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least 5 key foraging areas within Puerto Rico, USVI, and Florida.

Although the most recent 5-year review indicates there is not enough information to evaluate the statistical significance of nesting trends, nesting populations are increasing at the Puerto Rico (Mona Island) and U.S. Virgin Islands (Buck Island Reef National Monument) index beaches. Also in the U.S. Caribbean, additional nesting beaches are now being more systematically monitored to allow for future population trend assessments. Elsewhere in the Caribbean outside U.S. jurisdiction, nesting populations in Antigua/Barbuda and Barbados are increasing; however, other important nesting concentrations in the insular Caribbean are decreasing or their status is unknown, including Antigua/Barbuda (except Jumby Bay), Bahamas, Cuba (Doce Leguas Cays), Jamaica, and Trinidad and Tobago (NMFS 2013b).

The status of adults, subadults, and juveniles on foraging grounds is being monitored via in-water research. An in-water research project at Mona Island, Puerto Rico, has been ongoing for 15 years. However, abundance indices have not yet been incorporated into a rigorous analysis or a published trends assessment, as of yet. In addition, standardized in-water surveys have been initiated within the wider Caribbean (e.g., Pearl Cays, Nicaragua), but the time series is not long enough to detect a trend. In Florida, 2 in-water projects have been ongoing in Key West and Marquesas Keys conducted by the In-Water Research Group and Palm Beach County (NMFS 2013b).

The proposed action could cause the loss of up to 1 hawksbill sea turtles every 3 years and the animals may or may not be adult and may or may not be female. Our evaluation of potential future mortality is based on our belief that the same level of interactions occurred in the past, and with that level we have still seen positive trends in the status of this species. We determined the potential lethal captures associated with the proposed action would not have any detectable influence on the magnitude of the current nesting trends. While information on trends for adults, subadults, and juveniles at key foraging areas is not yet available, we also believe it is unlikely the potential removal of 1 hawksbills every 3 years will have any detectable influence over the numbers of adults, subadults, and juveniles occurring at 5 key foraging areas. Thus, we believe the proposed action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the hawksbill sea turtle in the wild.

7.2 Smalltooth Sawfish- U.S. DPS

The proposed action may result in up to 23 smalltooth sawfish takes (22 non-lethal, 1 lethal), every 3 years. The non-lethal takes of up to 22 smalltooth sawfish every three years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these takes may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of smalltooth sawfish is anticipated.

The loss of 1 smalltooth sawfish every 3 years will reduce the number of smalltooth sawfish as compared to the number of smalltooth sawfish that would have been present in the absence of the proposed action assuming all other variables remained the same. These lethal takes could also result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed action, if a female is taken. An adult female smalltooth sawfish may have a litter of approximately 10 pups probably every two years; therefore, the loss of one adult female smalltooth sawfish, could preclude the production of these pups. Because smalltooth sawfish produce more well-developed young it is likely that some portion of these pups would have survived. Thus, the death of a female eliminates an individual's contribution to future generations, and the proposed action would result in a reduction in future smalltooth sawfish reproduction. The loss of one animal from the population every 3 years will have no impact on the distribution of the species.

While there is currently no accurate smalltooth sawfish population estimate, a trend analysis of their abundance in the Everglades National Park, considered within the species core range, shows a slightly increasing population abundance trend since 1972 (Carlson et al. 2007). From 1989-2004, smalltooth sawfish relative abundance has increased 5% annually (Carlson and Osborne 2012; NMFS 2010). Using a demographic approach and life history data from similar species, Simpfendorfer (2000) estimates the most likely range for the intrinsic rate of increase is 0.08 per year to 0.13 per year with population doubling times of 10.3 to 13.5 years. Although this rate is very slow, the lethal take of one adult smalltooth sawfish every 3 years is not expected to have any measureable impact on this rate of population doubling-time. Even with the ongoing fishing activities associated with the proposed action, the smalltooth sawfish population still remains stable or increasing (Carlson and Osborne 2012). Although the anticipated mortality of three smalltooth sawfish every three years would result in an instantaneous reduction in absolute population number, we do not believe these mortalities will have any measurable effect on these trends. Therefore, we believe the anticipated lethal and non-lethal take of smalltooth sawfish associated with the proposed action are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the species in the wild.

Although we believe no change in distribution of smalltooth sawfish will occur as a result of the proposed action, we concluded the lethal take would result in an instantaneous reduction in absolute population numbers that may also reduce reproduction, but the short-term reductions are not expected to appreciably reduce the likelihood of survival of the

species in the wild. The following analysis considers the effects of that take on the likelihood of recovery in the wild. We consider the recovery objectives in the recovery plan prepared for the species that relate to population numbers or reproduction that may be affected by the predicted reductions in the numbers or reproduction of smalltooth sawfish resulting from the proposed action.

The recovery plan for the smalltooth sawfish (NMFS 2009) lists three main objectives as recovery criteria for the species. The two objectives and the associated sub-objectives relevant to the proposed action are:

Objective - Minimize Human Interactions and Associated Injury and Mortality

Sub-objective:

- Minimize human interactions and resulting injury and mortality of smalltooth sawfish through public education and outreach targeted at groups that are most likely to interact with sawfish (e.g., fishermen, divers, boaters).
- Develop and seek adoption of guidelines for safe handling and release of smalltooth sawfish to reduce injury and mortality associated with fishing.
- Minimize injury and mortality in all commercial and recreational fisheries.

Objective - Ensure Smalltooth Sawfish Abundance Increases Substantially and the Species Reoccupies Areas From Which it had Previously Been Extirpated

Sub-objective:

- Sufficient numbers of juvenile smalltooth sawfish inhabit several nursery areas across a diverse geographic area to ensure survivorship and growth and to protect against the negative effects of stochastic events within parts of their range.
- Adult smalltooth sawfish (> 340 cm) are distributed throughout the historic core of the species' range (both the Gulf of Mexico and Atlantic coasts of Florida). Numbers of adult smalltooth sawfish in both the Atlantic Ocean and Gulf of Mexico are sufficiently large that there is no significant risk of extirpation (i.e., local extinction) on either coast.
- Historic occurrence and/or seasonal migration of adult smalltooth sawfish are reestablished or maintained both along the Florida peninsula into the South-Atlantic Bight, and west of Florida into the northern and/or western Gulf of Mexico.

NMFS is currently funding several actions identified in the Recovery Plan for smalltooth sawfish; adult satellite tagging studies, the NSED, and monitoring take in commercial fisheries. Additionally, NMFS has developed safe-handling guidelines for the species. Despite the ongoing threats from the proposed action, we have still seen a stable or slightly increasing trend in the status of this species. Thus, the proposed action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the

likelihood of the U.S. DPS of smalltooth sawfish's recovery in the wild. NMFS must continue to monitor the status of the population to ensure the species continues to recover. Non-lethal takes of smalltooth sawfish will not affect the population of reproductive adult females. The potential lethal take of one smalltooth sawfish every 3 years will result in a reduction in overall population numbers in any given year. We have already determined that while these takes would likely result in an instantaneous reduction in absolute population numbers, we do not believe those reductions will have any measurably effects on the species increasing population trends. Additionally, we believe the proposed action will not impede the achievement of the relevant recovery objectives or sub-objectives. The HMS fisheries do not occur in areas currently believed to be juvenile nursery areas. The loss of one smalltooth sawfish every 3 years is not likely to have any discernible effect on the distribution of smalltooth sawfish or the ability for the species to re-establish its historical occurrence or seasonal migrations. Thus, the effects of the proposed action will not result in an appreciable reduction in the likelihood of smalltooth sawfish recovery in the wild.

Conclusion

The effects from proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of smalltooth sawfish in the wild.

7.3 Atlantic Sturgeon

Five DPSs of Atlantic sturgeon have been listed, 4 as endangered and 1 as threatened. Because Atlantic sturgeon mix extensively in the marine range, individuals from all 5 DPSs could occur in the action area, and are likely to be adversely affected by the proposed action. Therefore, a jeopardy determination must be made for each Atlantic sturgeon DPS and would be reached if the proposed action would appreciably reduce the likelihood of survival and recovery of any of the DPSs.

7.3.1 Gulf of Maine DPS

The proposed action may result in 34 Atlantic sturgeon takes from the Gulf of Maine (GOM) DPS every 3 years. We estimate those takes would be 9 adults (7 non-lethal, 2 lethal take) and 25 subadults (19 non-lethal, 6 lethal take).

The potential non-lethal takes of 26 (7 adults, 19 subadults) Atlantic sturgeon every 3 years are not expected to have any measurable impact on the reproduction, numbers, or distribution of animals from the GOM DPS. The individuals are expected to fully recover such that no reductions in reproduction or numbers of Atlantic sturgeon are anticipated.

Atlantic sturgeon travel extensively throughout the marine environment and have large ranges over which they disperse. Because the anticipated takes (both lethal and nonlethal) could occur anywhere within the range of the species, no change in the distribution of the GOM DPS Atlantic sturgeon is anticipated.

The potential lethal take would reduce the population of Atlantic sturgeon in the GOM DPS by 8 Atlantic sturgeon (2 adult, 6 subadult) over consecutive 3-year periods. Adult

Atlantic sturgeon are generally considered more important to the species because of their ability to breed. For this reason, we believe the best way to evaluate the impacts of the proposed action on Atlantic sturgeon reproduction is to consider not only how it is likely to affect adults, but also how it would affect subadults that may have lived to become adults (“adult equivalents”). GARFO-PRD developed an approach for estimating “adult equivalents” detailed previously, determining that each subadult equates to 0.48 adults. By multiplying that value by our estimates of subadult takes for each DPS from Section 5 we calculated the likely number of adult equivalents that may be captured during HMS fisheries activities. For the GOM DPS, we anticipate 3 adult equivalents may be killed every 3 years during the proposed action.⁴⁷ Therefore, we anticipate the proposed action is likely to result in 5 (2 adult, 3 adult equivalent) lethal adult/adult equivalent Atlantic sturgeon takes over consecutive 3-year periods from the GOM DPS. We will conduct this same conversion exercise for each subsequent DPS.

For the population of GOM DPS Atlantic sturgeon to remain stable over generations, a certain amount of spawning must occur across the entire DPS to offset deaths within the population. Two ways to measure spawning potential 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 during its lifetime to the total weight of the fish in a stock that is old enough to spawn, 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 1997a). 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 (1997a) indicates that since stock biomass and egg production are typically linearly correlated (i.e., larger individuals generally produce more eggs than smaller individuals) it is appropriate to apply the 20% (Goodyear 1993) threshold directly to EPR estimates.

Boreman (1997a) 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}). We believe evaluating the potential effects of the proposed action against the fishing mortality associated ($F = 0.14$) with maintaining an EPR of at least 20% of EPR_{max} is appropriate for evaluating the potential impacts of the proposed action on the likelihood the GOM DPS will survive in the wild.

Other Opinions have also considered the effects from other federal fisheries on Atlantic sturgeon. Likewise, a quantitative estimate of current/future Atlantic sturgeon takes exists for the American shad fishery in Georgia and North Carolina’s inshore gillnet fishery. Our

⁴⁷ 6 lethal GOM subadult takes x 0.48 subadult survival = 2.9 adult equivalents

analysis will include the authorized/calculated takes reported in the federal Biological Opinions as well as the Georgia and North Carolina fisheries since our analysis uses a published literature standard ($F=0.14=EPR_{20\%}$) that includes known fishing mortality from all fishing sources (i.e., federal and state fisheries). The GARFO batched consultation on 7 FMPs (NMFS 2013a) also determined up to 22 Atlantic sturgeon adult/adult equivalents would be lethally taken annually from the GOM DPS. The incidental take of Atlantic sturgeon in the commercial shrimp fishery of the South Atlantic (NMFS 2014b) estimated 1 Atlantic sturgeon from the GOM DPS would be killed annually.

NMFS completed a Programmatic Consultation on the Continued Prosecution of Fisheries and Ecosystem Research Conducted and Funded by the Northeast Fisheries Science Center that estimated 0.6 Atlantic sturgeon from the GOM DPS would be killed annually.

The Incidental Take Permit (ITP) (No. 16645) provided to Georgia in response to their Section 10 application provides for up to 0.52 lethal takes of Atlantic sturgeon from the GOM DPS over the course their 10 year permit (2.6 per year all DPSs x .02 from GOM DPS = 0.052 annually) and the Opinion analyzing those takes indicates those takes will be juveniles and subadults (NMFS 2013c). Converting those animals to adult equivalents as done previously decreases the number further, but not to zero.⁴⁸ Therefore, to be conservative for the species, we will assume the 0.052 animal potentially taken annually would have survived to be an adult and will consider it an adult equivalent.

The ITP (No. 18102) provided to North Carolina in response to their Section 10 application provides for up to 7 lethal takes of Atlantic sturgeon annually through 2023. The Opinion issuing those takes indicates those takes will be juveniles and subadults (NMFS 2014d). Following the previously discussed process for estimating the adult equivalents, we will consider 4 of those captures as adult equivalents.⁴⁹

Each year the SEFSC, state resource management agencies, USFWS, and academic institutions receive funding support from NMFS to collect fisheries independent data. This suite of independent but related activities collectively makes up NMFS's integrated fisheries independent monitoring (FIM) activities in the Southeast Region. Up to 0.6 adult animals from this DPS are expected to be lethally taken annually from these activities.

The 2017 Biological Opinion on the USFWS Funding of GCRD to Collect, Analyze and Report Biological and Fisheries Information to Describe the Conditions or Health of Recreationally Important Finfish was completed in January, 2017 (NMFS 2017a). Up to 0.4 adults from this DPS are expected to be lethally taken annually from these activities.

Together, Opinions for the GARFO batched FMP, the NEFSC, the Southeast shrimp trawl fishery, the Georgia shad fishery, the North Carolina gillnet fisheries, the USFWS/GCRD consultation, and the proposed action are estimated to result in 31.3 GOM DPS adult/adult equivalent mortalities annually. The NEAMAP model referenced earlier in this Opinion

⁴⁸ $0.052 \text{ annual juvenile/subadult Georgia shad gillnet takes} \times 0.48 \text{ subadult survival} = 0.025 \text{ adult equivalents}$

⁴⁹ $7 \text{ annual juvenile/subadult North Carolina gillnet takes} \times 0.48 \text{ subadult survival} = 3.36 \text{ adult equivalents}$

estimates a minimum ocean population of 7,455 Atlantic sturgeon in the GOM DPS, of which 4,548 are adults/subadults (Table 7.1). Therefore, our anticipated lethal takes represent 0.69% of the adult/adult equivalent population in the GOM DPS.⁵⁰ This is below the estimated 14% fishing mortality rate we believe the population could likely withstand and still maintain $EPR_{20\%}$. Based on this information, we believe the proposed action's removal of up to 5 adults/adult equivalents over 3 years (1.7 annually) will cause a reduction in numbers and reproduction. However, we do not believe these reductions will appreciably reduce the likelihood that the GOM DPS will survive in the wild.

Table 7.1 Calculated Ocean Population Estimates with Adult Equivalents (A.E.)

DPS	Estimated Ocean Population	Estimated Adult Ocean Population	Estimated Subadult Ocean Population*	Estimated Ocean Population of A.E.**	Estimated Ocean Population of Adults/A.E.
GOM	7,455	1,864	5,591	2,684	4,548
NYB	34,566	8,642	25,925	12,444	21,086
CB	8,811	2,203	6,608	3,172	5,375
Carolina	1,356	339	1,017	488	827
SA	14,911	3,728	11,183	5,368	9,096

*This estimate reflects the animals of a size vulnerable to capture in fisheries.

**This column estimated by multiplying the subadult population from previous column by 0.48.

Recovery

Our analysis must also consider whether the proposed action is likely to impede the recovery of Atlantic sturgeon from the GOM DPS. Because the GOM DPS of Atlantic sturgeon has only recently been listed, a recovery plan for this segment of the population has not yet been developed. However, a key step in recovering a species is to reduce threats identified as contributing to a species' threatened or endangered status; only by alleviating these threats can lasting recovery be achieved.

The final listing rule noted several major threats affecting the GOM DPS:

- 1) Dredging that can displace sturgeon while it is occurring and affect the quality of the habitat afterwards by changing the depth, sediment characteristics, and prey availability.
- 2) Degraded water quality in areas as a result of withdrawals for public use, runoff from agriculture, industrial discharges, and the alteration of river systems by dams and reservoirs.
- 3) Impeded access to historical habitat by dams and reservoirs.
- 4) Bycatch of Atlantic sturgeon in commercial fisheries.

⁵⁰ (1 Shrimp fishery take + 22 GARFO batched fisheries takes + 0.6 NEFSC research, + 4 North Carolina gillnet fisheries + 1 Georgia shad fishery + 0.6 FIM research + 0.4 USFWS/GCRD + 1.7 takes from the proposed action) ÷ 4,548 estimated adults/adult equivalents in the GOM DPS = 0.69% of the GOM DPS taken

- 5) Inadequacy of regulatory mechanisms to control bycatch and the modification and curtailment of Atlantic sturgeon habitat.

Nothing about the proposed action will affect the habitat or water quality or curtail the range of the species in the GOM DPS, and no dredging is involved. The proposed action has no relationship to the blockage of access to historical habitats by dams or reservoirs. The proposed action will not have negative impacts on the issue of regulatory mechanisms regarding control of bycatch and the modification and curtailment of Atlantic sturgeon habitat. The bycatch of Atlantic sturgeon in fishing gear will occur under the proposed action. However, we anticipate primarily nonlethal incidental captures that will be documented and procedures have been established to minimize the impact of any interactions that do occur. For these reasons, we believe the proposed action is not likely to appreciably reduce the likelihood that the GOM DPS will recover in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the GOM DPS of Atlantic sturgeon in the wild.

7.3.2 New York Bight DPS

The proposed action may result in 170 Atlantic sturgeon takes from the New York Bight (NYB) DPS over 3 years. We estimate those takes would be up to 43 adults (34 non-lethal, 9 lethal) and 127 subadults (100 non-lethal, 27 lethal).

The potential non-lethal takes of 134 Atlantic sturgeon every 3 years (34 adults, 100 subadults) are not expected to have any measurable impact on the reproduction, numbers, or distribution of these animals from the NYB DPS. The individuals are expected to fully recover such that no reductions in reproduction or numbers of Atlantic sturgeon are anticipated.

Atlantic sturgeon travel extensively throughout the marine environment and have large ranges over which they disperse. Because the anticipated takes (both lethal and non-lethal) could occur anywhere within the range of the species, no change in the distribution of the NYB DPS Atlantic sturgeon is anticipated.

The potential lethal take of 36 Atlantic sturgeon every 3 years (9 adults, 27 subadults) would reduce the population of Atlantic sturgeon in the NYB DPS by that amount. Because of the importance of breeding adults to a population, we will use the same approach described above and consider the proposed actions likely effects on subadults that may have lived to become adults (“adult equivalents”). Based on those calculations, we estimated the number of adult equivalents for the NYB DPS lethally taken by the proposed action will be 13 over 3 years.⁵¹ Thus, we anticipate the proposed action is likely to result in 22 adult/adult equivalent Atlantic sturgeon (9 adults, 13 adult equivalents) lethal takes over 3 years (7.3 annually) from the NYB DPS.

⁵¹ 27 lethal NYB subadult takes x 0.48 subadult survival = 12.96 adult equivalents

To determine whether that reduction would appreciably reduce the species' likelihood of survival in the wild we will follow the same approach and assumptions we discussed previously in Section 7.5.1. We will evaluate those takes relative to the 14% fishing mortality rate Boreman (1997a) reported adult female Atlantic sturgeon in the Hudson River could have likely sustained and still retained enough spawners for the population to remain stable (i.e., maintain an EPR of at least 20% of EPR_{max}).

We anticipated 22 adult/adult equivalents may be lethally taken by the proposed action over 3 years (7.3 annually). Additionally, we anticipate lethal NYB DPS takes in the Southeastern shrimp fishery (3 annually) (NMFS 2012e), the 7 fisheries analyzed in the GARFO batched consultation (100 annually) (NMFS 2013a) and the NEFSC programmatic consultation (NMFS 2016) (3 annually).

The Georgia ITP provides for up to 2.6 lethal takes of Atlantic sturgeon from the NYB DPS over the course their 10 year permit (0.26 annually); indicating those takes will be juveniles and subadults (NMFS 2013c). Converting those animals to adult equivalents as done previously yields a number less than 1, but not zero.⁵² To be conservative for the species, we will assume the 0.26 animal potentially taken annually would have survived to be an adult and will consider it an adult equivalent.

The ITP (No. 18102) provided to North Carolina provides for up to 18 lethal takes of Atlantic sturgeon from the NYB DPS annually through 2023. The Opinion issuing those takes indicates those takes will be juveniles and subadults (NMFS 2014d). Following the previously discussed process for estimating the adult equivalents, we will consider 9 of those captures as adult equivalents.⁵³

Up to 1 adult animal from this DPS is expected to be lethally taken annually from NMFS's FIM activities in the Southeast Region.

The USFWS Funding of GCRD anticipated 2 adult/adult equivalents from this DPS may be lethally taken by the proposed action over 5 years (0.4 annually).

We anticipate that 124.7 adult/adult equivalent Atlantic sturgeon may be taken annually in these fisheries and by the proposed action. The NEAMAP model estimates a minimum ocean population of 34,556 Atlantic sturgeon in the NYB DPS, of which 21,086 are adults/subadults (Table 7.1). Based on this information, we believe 0.59% of the adult/adult equivalent population in the NYB DPS will be killed annually.⁵⁴ This 0.59% is below the estimated 14% total fishing mortality rate we believe the population could likely withstand and still maintain $EPR_{20\%}$. Based on this information, we believe the proposed

⁵² 0.26 annual juvenile/subadult Georgia shad gillnet takes x 0.48 subadult survival = 0.13 adult equivalents

⁵³ 18 annual juvenile/subadult North Carolina gillnet takes x 0.48 subadult survival = 8.64 adult equivalents

⁵⁴ (3 Shrimp fishery takes + 100 GARFO batched fisheries takes + 3 NEFSC + 1 Georgia shad fishery + 9 North Carolina gillnet fisheries + 1 FIM research + 0.4 USFWS/GCRD + 7.3 estimated takes from the proposed action) ÷ 21,086 estimated adults/adult equivalents in the NYB DPS = 0.59% of the NYB DPS taken

action's removal of up to 7.3 adults/adult equivalents over 3-years will cause a reduction in numbers and reproduction. However, we do not believe these reductions are likely to cause an appreciable reduction in the likelihood that the NYB DPS will survive in the wild.

Recovery

Our analysis must also consider whether the proposed action is likely to impede the recovery of Atlantic sturgeon from this DPS. Because this DPS of Atlantic sturgeon has only recently been listed, a recovery plan for this segment of the population has not yet been developed. However, a key step in recovering a species is to reduce threats identified as contributing to a species' threatened or endangered status; only by alleviating these threats can lasting recovery be achieved.

The final listing rule noted several major threats affecting Atlantic sturgeon in the NYB DPS:

- 1) Dredging that can displace sturgeon while it is occurring and affect the quality of the habitat afterwards by changing the depth, sediment characteristics, and prey availability.
- 2) Degraded water quality in areas throughout the range of the five DPSs as a result of withdrawals for public use, runoff from agriculture, industrial discharges, and the alteration of river systems by dams and reservoirs.
- 3) Impeded access to historical habitat by dams and reservoirs.
- 4) Bycatch of Atlantic sturgeon in commercial fisheries.
- 5) Vessel strikes within the riverine portions of the range of the New York Bight.
- 6) Inadequacy of regulatory mechanisms to control bycatch and the modification and curtailment of Atlantic sturgeon habitat.

No dredging is involved with the proposed action. Nothing about the proposed action will affect the habitat or water quality or curtail the range of the species, in the NYB DPS. The proposed action has no relationship to the blockage of access to historical habitats by dams or reservoirs.

The proposed action could introduce threats of vessel strikes. We believe the threats from vessel strikes to the NYB DPS of Atlantic sturgeon are not of concern when considering the potential effect from this threat to the recovery of the NYB DPS. Given the lack of any previous documented interactions with HMS fishing vessels, the types of vessels, and monitoring for protected species anytime the vessel is moving, this Opinion found that adverse effects from vessel operations are extremely unlikely to occur.

The bycatch of Atlantic sturgeon in fishing gear will occur under the proposed action. However, we anticipate primarily nonlethal incidental captures that will be documented and procedures have been established to minimize the impact of any interactions that do occur. The proposed action will not have negative impacts on the issue of regulatory mechanisms regarding control of bycatch and the modification and curtailment of Atlantic

sturgeon habitat. For these reasons, we believe the proposed action is not likely to appreciably reduce the likelihood that the NYB DPS will recover in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NYB DPS of Atlantic sturgeon in the wild.

7.3.3 Chesapeake Bay DPS

The proposed action may result in 40 Atlantic sturgeon takes from the Chesapeake Bay (CB) DPS over 3-years. We estimate those takes would be up to 10 adults (7 non-lethal, 3 lethal) and 30 subadults (24 non-lethal, 6 lethal).

The potential non-lethal takes of 31 Atlantic sturgeon annually (7 adults, 24 subadults) are not expected to have any measurable impact on the reproduction, numbers, or distribution of these animals from the CB DPS. The individuals are expected to fully recover such that no reductions in reproduction or numbers of Atlantic sturgeon are anticipated.

Atlantic sturgeon travel extensively throughout the marine environment and have large ranges over which they disperse. Because the anticipated takes (both lethal and nonlethal) could occur anywhere within the range of the species, no change in the distribution of the CB DPS Atlantic sturgeon is anticipated.

The potential lethal take would reduce the population of Atlantic sturgeon in the CB DPS. As discussed previously, we believe breeding adults are especially important to the overall populations of the Atlantic sturgeon DPSs. For that reason, we followed the same approach described in Section 7.5.1 to estimate adult equivalents for the CB DPS. Based on those calculations we estimated the number of adult equivalents for the CB DPS affected by the proposed action was 3 over 3-years.⁵⁵ Thus, we anticipate the proposed action is likely to result in 6 Atlantic sturgeon (3 adult, 3 adult equivalent) lethal takes over 3-years from the CB DPS (2 annually).

To determine whether that reduction would appreciably reduce the species' likelihood of survival in the wild, we will follow the same approach and assumptions we discussed previously in Section 7.5.1. We will evaluate those takes relative to the 14% fishing mortality rate Boreman (1997a) reported adult female Atlantic sturgeon in the Hudson River could have likely sustained and still retained enough spawners for the population to remain stable (i.e., maintain an EPR of at least 20% of EPR_{max}).

We anticipated 6 adult/adult equivalents may be taken by the proposed action over consecutive 3-year periods (2 annually). Additionally, we anticipate lethal CB DPS takes in the Southeastern shrimp fishery (2 annually) (NMFS 2012e), the 7 fisheries analyzed in the GARFO batched consultation (27 annually) (NMFS 2013a), and the NEFSC programmatic consultation (NMFS 2016) (0.8 annually).

⁵⁵ 6 lethal CB subadult takes x 0.48 subadult survival = 2.9 adult equivalents

The Georgia ITP provides for up to 5.2 lethal takes of Atlantic sturgeon from the CB DPS over the course their 10 year permits (0.52 annually); indicating those takes will be juveniles and subadults (NMFS 2013c). Converting those animals to adult equivalents as done previously yields a number less than 1, but not zero.⁵⁶ To be conservative for the species, we will assume the 0.52 animal potentially taken annually would have survived to be an adult and will consider it an adult equivalent.

The North Carolina ITP (No. 18102) provides for up to 69 lethal takes of Atlantic sturgeon from the CB DPS annually through 2023. The Opinion issuing those takes indicates those takes will be juveniles and subadults (NMFS 2014d). Following the previously discussed process for estimating the adult equivalents, we will consider 33 of those captures as adult equivalents.⁵⁷

Up to 0.6 adult animal from this DPS are expected to be lethally taken annually from NMFS FIM activities in the Southeast Region.

The USFWS funding of GCRD anticipated 0.4 lethal takes of adult/adult equivalents from this DPS annually.

We anticipate that 67 adult Atlantic sturgeon may be taken annually in these fisheries and by our proposed action. The NEAMAP model estimates a minimum ocean population of 8,811 Atlantic sturgeon in the CB DPS, of which 5,375 are adults/subadults (Table 7.1). Based on this information, we believe 1.25% of the adult/adult equivalent population in the CB DPS will be killed annually.⁵⁸ This 1.25% is below the estimated 14% total fishing mortality rate we believe the population could likely withstand and still maintain EPR_{20%}. Based on this information, we believe the proposed action's removal of up to 6 adult/adult equivalent over 3 years will cause a reduction in numbers and reproduction. However, we do not believe these reductions are likely to cause an appreciable reduction in the likelihood that the CB DPS will survive in the wild.

Recovery

Our analysis must also consider whether the proposed action is likely to impede the recovery of Atlantic sturgeon from this DPS. Because this DPS of Atlantic sturgeon has only recently been listed, a recovery plan for this segment of the population has not yet been developed. However, a key step in recovering a species is to reduce threats identified as contributing to a species' threatened or endangered status; only by alleviating these threats can lasting recovery be achieved.

The final listing rule noted several major threats affecting Atlantic sturgeon in the CB DPS:

⁵⁶ 0.52 annual juvenile/subadult Georgia shad gillnet takes x 0.48 subadult survival = 0.25 adult equivalents

⁵⁷ 69 annual juvenile/subadult North Carolina gillnet takes x 0.48 subadult survival = 33 adult equivalents

⁵⁸ (2 Shrimp fishery takes + 27 GARFO batched fisheries takes + 0.8 NEFSC takes + 1 Georgia shad fishery + 33 North Carolina fisheries + 0.6 FIM + 0.4 USFWS/GCRD + 2 estimated takes from the proposed action) ÷ 5,375 estimated adults/adult equivalents in the CB DPS = 1.25% of the CB DPS taken.

- 1) Dredging that can displace sturgeon while it is occurring and affect the quality of the habitat afterwards by changing the depth, sediment characteristics, and prey availability.
- 2) Degraded water quality in areas throughout the range of the 5 DPSs as a result of withdrawals for public use, runoff from agriculture, industrial discharges, and the alteration of river systems by dams and reservoirs.
- 3) Bycatch of Atlantic sturgeon in commercial fisheries.
- 4) Vessel strikes in within the riverine portions of the range of CB DPS.
- 5) Inadequacy of regulatory mechanisms to control bycatch and the modification and curtailment of Atlantic sturgeon habitat.

No dredging is involved with the proposed action. Nothing about the proposed action will affect the habitat or water quality or curtail the range of the species, in the CB DPS. The proposed action could introduce threats of vessel strikes and bycatch from fishing gear. However, given the lack of any previous documented interactions with HMS fishing vessels, the types of vessels, and monitoring for protected species anytime the vessel is moving, this Opinion found that adverse effects from vessel operations are extremely unlikely to occur. Therefore, we believe the threats from vessel strikes to the CB DPS of Atlantic sturgeon are not of concern when considering the potential effect from this threat to the recovery of the CB DPS. The proposed action will not have negative impacts on the issue of regulatory mechanisms regarding control of bycatch and the modification and curtailment of Atlantic sturgeon habitat. The bycatch of Atlantic sturgeon in fishing gear will occur under the proposed action. However, we anticipate primarily nonlethal incidental captures that will be documented and procedures have been established to minimize the impact of any interactions that do occur. For these reasons, we believe the proposed action is not likely to appreciably reduce the likelihood that the CB DPS will recover in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the CB DPS of Atlantic sturgeon in the wild.

7.3.4 South Atlantic DPS

The proposed action may result in 75 Atlantic sturgeon takes from the South Atlantic (SA) DPS over 3 years. We estimate those takes would be up to 21 adults (14 non-lethal, 7 lethal) and 54 subadults (42 non-lethal, 12 lethal).

The potential non-lethal takes of 56 Atlantic sturgeon every 3 years (14 adults, 42 subadults) are not expected to have any measurable impact on the reproduction, numbers, or distribution of these animals from the SA DPS. The individuals are expected to fully recover such that no reductions in reproduction or numbers of Atlantic sturgeon are anticipated.

Atlantic sturgeon travel extensively throughout the marine environment and have large ranges over which they disperse. Because the anticipated takes (both lethal and non-lethal) could occur anywhere within the range of the species, no change in the distribution of the CB DPS Atlantic sturgeon is anticipated.

The potential lethal take of 19 Atlantic sturgeon annually (7 adults, 12 subadults) would reduce the population of Atlantic sturgeon in the SA DPS by that amount. Adult Atlantic sturgeon are generally considered more important to the species because of their ability to breed. For that reason, we followed the same approach described in Section 7.5.1 to estimate adult equivalents for the SA DPS. Based on those calculations we estimated the number of adult equivalents for the SA DPS affected by the proposed action was 6 over 3 years.⁵⁹ Thus, we anticipate the proposed action is likely to result in 13 Atlantic sturgeon (7 adults, 6 adult equivalents) lethal takes over 3 years from the SA DPS (4.3 annually).

To determine whether that reduction would appreciably reduce the species' likelihood of survival in the wild, we will follow the same approach and assumptions we discussed previously in Section 7.5.1. We will evaluate those takes relative to the 14% fishing mortality rate Boreman (1997a) reported adult female Atlantic sturgeon in the Hudson River could have likely sustained and still retained enough spawners for the population to remain stable (i.e., maintain an EPR of at least 20% of EPR_{max}).

We anticipated 13 (12.8 rounded to 13) adult/adult equivalents may be taken by the proposed action over 3 years (4.3 annually). Additionally, we anticipate lethal SA DPS takes in the Southeastern shrimp fishery (7 annually) (NMFS 2012e), the 7 fisheries analyzed in the GARFO batched consultation (43 annually) (NMFS 2013a), and the NEFSC programmatic consultation (1.4 annually).

The Georgia ITP provides for up to 24.7 lethal takes of Atlantic sturgeon from the SA DPS over the their 10 year permit (2.47 annually); indicating those takes will be juveniles and subadults (NMFS 2013c). Following the previously discussed process for estimating the adult equivalents, we will consider 2 of those captures as adult equivalents.⁶⁰

The North Carolina ITP (No. 18102) provides for up to 69 lethal takes of Atlantic sturgeon from the SA DPS annually through 2023. The Opinion issuing those takes indicates those takes will be juveniles and subadults (NMFS 2014d). Following the previously discussed process for estimating the adult equivalents, we will consider 33 of those captures as adult equivalents.⁶¹

Up to 0.8 adult animal from this DPS are expected to be lethally taken annually from NMFS FIM activities in the Southeast Region.

⁵⁹ 12 lethal SA subadult takes x 0.48 subadult survival = 5.8 adult equivalents

⁶⁰ 2.47 annual juvenile/subadult Georgia shad gillnet takes x 0.48 subadult survival = 1.19 (round to 2) adult equivalents

⁶¹ 69 annual juvenile/subadult North Carolina gillnet takes x 0.48 subadult survival = 33 adult equivalents

The consultation on the USFWS funding of GCRD anticipated lethal take of 0.4 adults/adult equivalents from this DPS annually.

We anticipate that 91.9 adult Atlantic sturgeon from the SA DPS may be taken annually in these fisheries and by our proposed action. The NEAMAP model estimates a minimum ocean population of 14,911 Atlantic sturgeon in the SA DPS, of which 9,096 are adults/subadults (Table 7.1). Based on this information, we believe 1% of the adult/adult equivalent population in the SA DPS will be killed annually.⁶² This 1% is below the estimated 14% total fishing mortality rate we believe the population could likely withstand and still maintain EPR20%. Based on this information, we believe the proposed action's removal of up to 13 adult/adult equivalent over 3 years will cause a reduction in numbers and reproduction. However, we do not believe these reductions are likely to cause an appreciable reduction in the likelihood that the SA DPS will survive in the wild.

Recovery

Our analysis must also consider whether the proposed action is likely to impede the recovery of Atlantic sturgeon from this DPS. Because this DPS of Atlantic sturgeon has only recently been listed, a recovery plan for this segment of the population has not yet been developed. However, a key step in recovering a species is to reduce threats identified as contributing to a species' threatened or endangered status; only by alleviating these threats can lasting recovery be achieved.

The final listing rule noted several major threats affecting the SA DPS:

- 1) Dredging that can displace sturgeon while it is occurring and affect the quality of the habitat afterwards by changing the depth, sediment characteristics, and prey availability.
- 2) Degraded water quality in areas as a result of withdrawals for public use, runoff from agriculture, industrial discharges, and the alteration of river systems by dams and reservoirs.
- 3) Impeded access to historical habitat by dams and reservoirs.
- 4) Bycatch of Atlantic sturgeon in commercial fisheries.
- 5) Inadequacy of regulatory mechanisms to control bycatch and the modification and curtailment of Atlantic sturgeon habitat.

No dredging is involved with the proposed action. Nothing about the proposed action will affect the habitat or water quality or curtail the range of the species in the SA DPS. The proposed action has no relationship to the blockage of access to historical habitats by dams or reservoirs. The proposed action will not have negative impacts on the issue of regulatory mechanisms regarding control of bycatch and the modification and curtailment of Atlantic sturgeon habitat. The bycatch of Atlantic sturgeon in fishing gear will occur under the proposed action. However, we anticipate primarily nonlethal incidental captures

⁶²(7 Shrimp fishery takes + 43 GARFO batched fisheries takes + 1.4 NEFSC takes + 2 Georgia shad fishery + 33 North Carolina fisheries + 0.8 FIM + 0.4 USFWS/GCRD + 4.3 estimated takes from the proposed action) ÷ 9,096 estimated adults/adult equivalents in the SA DPS = 1% of the SA DPS taken.

that will be documented and procedures have been established to minimize the impact of any interactions that do occur. For these reasons, we believe the proposed action is not likely to appreciably reduce the likelihood that the SA DPS will recover in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of Atlantic sturgeon in the wild.

7.3.5 Carolina DPS

The proposed action may result in 10 Atlantic sturgeon takes from the Carolina DPS every 3 years. We estimate those takes would be 5 adult (1 non-lethal, 4 lethal) and 5 subadults (4 non-lethal, 1 lethal).

The potential non-lethal takes of 5 Atlantic sturgeon every 3 years (1 adult, 4 subadults) is not expected to have any measurable impact on the reproduction, numbers, or distribution of these animals from the Carolina DPS. The individuals are expected to fully recover such that no reductions in reproduction or numbers of Atlantic sturgeon are anticipated.

Atlantic sturgeon travel extensively throughout the marine environment and have large ranges over which they disperse. Because the anticipated takes (both lethal and non-lethal) could occur anywhere within the range of the species, no change in the distribution of the CB DPS Atlantic sturgeon is anticipated.

The potential lethal take of 5 lethal Atlantic sturgeon every 3 years (4 adults, 1 subadult) would reduce the population of Atlantic sturgeon in the Carolina DPS by that amount. Adult Atlantic sturgeon are generally considered more important to the species because of their ability to breed. For that reason, we followed the same approach described in Section 7.5.1 to estimate adult equivalents for the Carolina DPS.⁶³ Based on those calculations we estimated the number of adult equivalents for the Carolina DPS affected by the proposed action was 0.48 over 3 years. Thus, we anticipate the proposed action is likely to result in 5 Atlantic sturgeon (4 adults and 1 adult equivalent) lethal takes over consecutive 3-year periods from the Carolina DPS (1.7 annually).

To determine whether that reduction would appreciably reduce the species' likelihood of survival in the wild we will follow the same approach and assumptions we discussed previously in Section 7.5.1. We will evaluate those takes relative to the 14% fishing mortality rate Boreman (1997a) reported adult female Atlantic sturgeon in the Hudson River could have likely sustained and still retained enough spawners for the population to remain stable (i.e., maintain an EPR of at least 20% of EPR_{max}).

We anticipated 5 (4.48 rounded to 5) adult/adult equivalents may be taken by the proposed action over 3 years (1.7 annually). Additionally, we anticipate lethal Carolina DPS takes in the Southeastern shrimp fishery (3 annually) (NMFS 2012e), the 7 fisheries analyzed in the

⁶³ 1 lethal Carolina subadult takes x 0.48 subadult survival = 0.48 adult equivalents

GARFO batched consultation (5 annually) (NMFS 2013a), and the NEFSC programmatic consultation (0.2 annually).

The Georgia ITP provides for up to 3.9 lethal takes of Atlantic sturgeon from the Carolina DPS over the course their 10 year permits (0.39 annually); indicating those takes will be juveniles and subadults (NMFS 2013c). Converting those animals to adult equivalents as done previously yields a number less than 1, but not zero.⁶⁴ To be conservative for the species, we will assume the 0.39 animal potentially taken annually would have survived to be an adult and will consider it an adult equivalent.

The ITP (No. 18102) provided to North Carolina provides for up to 127 lethal takes of Atlantic sturgeon from the Carolina DPS annually through 2023. The Opinion issuing those takes indicates those takes will be juveniles and subadults (NMFS 2014d). Following the previously discussed process for estimating the adult equivalents, we will consider 61 of those captures as adult equivalents.⁶⁵

Up to 0.2 adult animal from this DPS are expected to be lethally taken annually from NMFS FIM activities in the Southeast Region.

The consultation on the USFWS funding of GCRD anticipated lethal take of 0.4 adults from this DPS annually.

We anticipate that 72.5 adult Atlantic sturgeon may be taken annually in these fisheries and by our proposed action. The NEAMAP model estimates a minimum ocean population of 1,356 Atlantic sturgeon in the Carolina DPS, of which 827 are adults/subadults (Table 7.1). Based on this information, we believe 8.8% of the adult/adult equivalent population in the Carolina DPS will be killed annually.⁶⁶ This 8.8% is below the estimated 14% total fishing mortality rate we believe the population could likely withstand and still maintain $EPR_{20\%}$. Based on this information, we believe the proposed action's removal of up to 5 adult/adult equivalent over 3 years will cause a reduction in numbers and reproduction. However, we do not believe these reductions are likely to cause an appreciable reduction in the likelihood that the Carolina DPS will survive in the wild.

Recovery

The final listing rule noted several major threats affecting the Carolina DPS:

- 1) Dredging that can displace sturgeon while it is occurring and affect the quality of the habitat afterwards by changing the depth, sediment characteristics, and prey availability.

⁶⁴ $0.39 \text{ annual juvenile/subadult Georgia shad gillnet takes} \times 0.48 \text{ subadult survival} = 0.19 \text{ adult equivalents}$

⁶⁵ $127 \text{ annual juvenile/subadult North Carolina gillnet takes} \times 0.48 \text{ subadult survival} = 61 \text{ adult equivalents}$

⁶⁶ $(3 \text{ Shrimp fishery takes} + 5 \text{ GARFO batched fisheries takes} + 0.2 \text{ NEFSC takes} + 1 \text{ Georgia shad fishery} + 61 \text{ North Carolina gillnet fisheries} + 0.2 \text{ FIM} + 0.4 \text{ USFWS/GCRD} + 1.7 \text{ estimated takes from the proposed action}) \div 827 \text{ estimated adults/adult equivalents in the Carolina DPS} = 8.8\% \text{ of the Carolina DPS taken}$

- 2) Degraded water quality in areas as a result of withdrawals for public use, runoff from agriculture, industrial discharges, and the alteration of river systems by dams and reservoirs.
- 3) Impeded access to historical habitat by dams and reservoirs.
- 4) Bycatch of Atlantic sturgeon in commercial fisheries.
- 5) Inadequacy of regulatory mechanisms to control bycatch and the modification and curtailment of Atlantic sturgeon habitat.

No dredging is involved with the proposed action. Nothing about the proposed action will affect the habitat or water quality or curtail the range of the species in the Carolina DPS. The proposed action has no relationship to the blockage of access to historical habitats by dams or reservoirs. The bycatch of Atlantic sturgeon in fishing gear will occur under the proposed action. However, we anticipate primarily nonlethal incidental captures that will be documented and procedures have been established to minimize the impact of any interactions that do occur. The proposed action will not have negative impacts on the issue of regulatory mechanisms regarding control of bycatch and the modification and curtailment of Atlantic sturgeon habitat. For these reasons, we believe the proposed action is not likely to appreciably reduce the likelihood that the Carolina DPS will recover in the wild.

Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the Carolina DPS of Atlantic sturgeon in the wild.

7.4 Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS

Only the Central and Southwest Atlantic DPS of scalloped hammerhead shark occurs within the action area, and is likely to be adversely affected by the proposed action; therefore, a jeopardy analysis must determine whether the proposed action will appreciably reduce the likelihood of survival and recovery of this DPS.

The proposed action may result in 7 (3 non-lethal and 4 lethal) scalloped hammerhead shark takes over consecutive 3-year periods. The nonlethal capture of 3 scalloped hammerhead sharks every 3 years, is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur only in the Caribbean area and would be released within the general area where caught, no change in the distribution of this species is anticipated.

We estimate that up to 4 of those takes may be lethal (1.3 annually). The loss of 4 scalloped hammerhead over consecutive 3-year periods will reduce the number of scalloped hammerhead as compared to the number of scalloped hammerhead that would have been present in the absence of the proposed action assuming all other variables remained the same. This lethal take could also result in the loss of reproduction value as

compared to the reproductive value in the absence of the proposed action, if a female is taken. Within the Central and Southwest Atlantic DPS, female scalloped hammerhead sharks have litters with between 2 and 21 pups, with an average of 14.3 (Hazin et al. 2001), probably every 2 years. While we have no reason to believe the proposed action will disproportionately affect females, the loss of 4 adult female scalloped hammerhead shark over consecutive 3-year periods (1.3 per year) could preclude the production of a maximum of 41 (10.5 pups per year*1.3 takes per year * 3 years, rounded up) pups every 3 years. Because scalloped hammerhead produce relatively well-developed young, it is likely that some portion of these pups would have survived. Thus, the death of a female eliminates an individual's contribution to future generations, and the proposed action would result in a reduction in future scalloped hammerhead reproduction. While scalloped hammerhead sharks are less migratory than other sharks, they are still wide-ranging. We believe the potential loss of 4 animals during consecutive 3-year periods would not affect the distribution of the species.

There is currently no accurate population estimate for the Central and Southwest Atlantic DPS of scalloped hammerhead sharks. However, Miller et al. (2014) concluded that abundance numbers for this DPS are likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS. The virgin population estimates for the Northwest Atlantic and Gulf of Mexico DPS ranged from 142,000 and 169,000 individuals (range 116,000-260,000) (Hayes et al. 2009). The population estimates for the most recent time period (2005) estimate a much smaller population: 24,850-27,900 individuals (Hayes et al. 2009). Since Miller et al. (2014) concluded that abundance numbers for this DPS are likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS, we will conservatively base our analysis on the 24,850 population number.

The lethal take of 4 scalloped hammerhead sharks every 3 years (1.3 annually) will reduce the number of scalloped hammerheads relative to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. This lethal take could also result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed action, if females were taken. However, we believe the loss of 4 scalloped hammerheads over consecutive 3-year periods will not significantly decrease the populations within the Central and Southwest Atlantic DPS as this is a very limited amount of loss nor will it change their distribution. Thus, we believe the proposed action is not likely to appreciably reduce the likelihood of survival of the Central and Southwest Atlantic DPS of scalloped hammerhead sharks in the wild.

The following analysis considers the effects of expected take on the likelihood of recovery in the wild. Since scalloped hammerhead sharks have just recently been listed, a recovery plan for them is not yet available. However, the first step in recovering a species is to reduce identified threats; only by alleviating threats can lasting recovery be achieved. The Final Listing Rule (79 FR 38213, July 3, 2014) noted the following potential threats to the Central and Southwest Atlantic DPS of scalloped hammerhead sharks:

- 1) Overutilization in artisanal fisheries, north of Brazil, that operate in nearshore and inshore environments that are likely nursery areas, and overutilization in artisanal and commercial fisheries within Brazil that target scalloped hammerhead sharks.
- 2) Operation of domestic artisanal fisheries and foreign commercial fisheries in areas without adequate fisheries regulations and operation of domestic and foreign fisheries in areas without capacity to enforce existing fishery regulations.
- 3) Scalloped hammerhead sharks' physiology makes them very susceptible to mortality in fishing gear. They often suffer very high at-vessel fishing mortality (e.g., Morgan and Burgess, 2007; Macbeth et al., 2009), and their schooling behavior increases their likelihood of being caught in large numbers.

Recovery is the process by which the ecosystems of the Central and Southwest Atlantic DPS of scalloped hammerhead sharks are restored and the threats to the species are removed. Restoring the ecosystem and eliminating threats will help support self-populating and self-regulating populations so they can become persistent members of the native biological communities (USFWS and NMFS 1998). As discussed previously, the proposed action is not likely to impede the Central and Southwest Atlantic DPS of scalloped hammerhead sharks from continuing to survive. The proposed action will not impede the process of restoring the ecosystems that affect the Central and Southwest Atlantic DPS of scalloped hammerhead sharks nor are these fisheries a significant threat (6 takes every three years). Thus, we believe the proposed action is not likely to impede the recovery of, and will not result in an appreciable reduction in, the likelihood of the Central and Southwest Atlantic DPS of scalloped hammerhead shark's recovery in the wild.

Conclusion

The effects from proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the Central and Southwest Atlantic DPS of scalloped hammerhead sharks in the wild.

7.5 Oceanic Whitetip Shark

The Oceanic whitetip shark occurs throughout the action area and is likely to be adversely affected by the proposed action; therefore, a jeopardy analysis must determine whether the proposed action will appreciably reduce the likelihood of survival and recovery of this species.

The proposed action may result in 6 oceanic whitetip shark takes over consecutive 3-year periods. We estimate that 3 of those takes may be nonlethal and 3 may be lethal (1 annually). The nonlethal capture of 3 Oceanic whitetip sharks every 3 years, is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur only anywhere in the action area and would be released within the general area where caught, no change in the distribution of this species is anticipated.

The loss of 3 oceanic whitetip sharks over consecutive 3-year periods will reduce the number of oceanic whitetip sharks as compared to the number of oceanic whitetip sharks that would have been present in the absence of the proposed action assuming all other variables remained the same. This lethal take could also result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed action, if a female is taken. Female oceanic whitetip shark have litters with between 1 and 14 pups, with an average of 6 probably every 2 years (Young et al. 2016). While we have no reason to believe the proposed action will disproportionately affect females, the loss of an adult female oceanic whitetip shark over consecutive 3-year periods could preclude the production of a maximum of 21 (7 pups per year*1 take per year * 3 years) pups every 3 years. Because oceanic whitetip sharks produce relatively well-developed young, it is likely that some portion of these pups would have survived. Thus, the death of a female eliminates an individual's contribution to future generations, and the proposed action would result in a reduction in future oceanic whitetip shark reproduction. We believe the potential loss of 3 animals during consecutive 3-year periods would not affect the distribution of the species.

There is currently no accurate population estimate for oceanic whitetip sharks. Oceanic whitetip sharks can be found worldwide, with no present indication of a range contraction. Oceanic whitetip sharks are wide-ranging. While a global population size estimate or trend for the oceanic whitetip shark is currently unavailable, numerous sources of information, including the results of a recent stock assessment and several other abundance indices are available to infer and assess current regional abundance trends of the species. Relative abundance of oceanic whitetip sharks may have stabilized in the North Atlantic since 2000 and in the Gulf of Mexico/Caribbean since the late 1990s at a significantly diminished abundance (Cortés et al. 2007; Young et al. 2016).

The lethal take of 3 oceanic whitetip sharks over consecutive 3-year periods (1 annually) will reduce the number of oceanic whitetip sharks relative to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. This lethal take could also result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed action, if females were taken. However, due to the small loss in numbers and reproductive value we believe the action will not result in population level impacts nor will it change their distribution. Thus, we believe the proposed action is not likely to appreciably reduce the likelihood of survival of the oceanic whitetip shark in the wild.

The following analysis considers the effects of expected take on the likelihood of recovery in the wild. Since oceanic whitetip sharks were recently listed, a recovery plan for them is not yet available. However, the first step in recovering a species is to reduce identified threats; only by alleviating threats can lasting recovery be achieved. The Final Listing Rule (83 FR 4153, January 30, 2018) noted the following potential threats to the oceanic whitetip shark: In the Northwest Atlantic, the oceanic whitetip is caught incidentally as bycatch by a number of fisheries, including (but not limited to) the U.S. Atlantic pelagic longline fishery (being analyzed in a separate opinion), the Cuban "sport" fishery ("sport" = private artisanal and commercial), and the Colombian oceanic industrial longline fishery

operating in the Caribbean. Oceanic whitetip sharks are also a preferred species for their large, morphologically distinct fins, as they obtain a high price in the Asian fin market, and thus they are valuable as incidental catch for the international shark fin trade. Oceanic whitetip sharks possess life history characteristics that increase their vulnerability to harvest, including slow growth, relatively late age of maturity, and low fecundity. The species' low genetic diversity in concert with steep global abundance declines and ongoing threats of overutilization may pose a viable risk to the species in the foreseeable future.

The final rule also noted that the potential stabilization of oceanic whitetip sharks in the proposed action area occurred concomitantly with the first FMP for Sharks in the Northwest Atlantic Ocean and Gulf of Mexico described in Section 2 of this Opinion. Oceanic whitetips sharks are managed directly under the pelagic shark group, and the FMP has included regulations on trip limits and quotas. This indicates the potential efficiency of these management measures for reducing the threat of overutilization of the oceanic whitetip shark population in this region.

Recovery is the process by which the ecosystems of oceanic whitetip sharks are restored and the threats to the species are removed. Restoring ecosystems and eliminating threats will support self-populating and self-regulating populations so they can become persistent members of the native biological communities (USFWS and NMFS 1998). As discussed previously, the proposed action is not likely to impede oceanic whitetip sharks from continuing to survive. The proposed action will not impede the process of restoring the ecosystems that affect oceanic whitetip sharks nor are these fisheries a significant threat (3 lethal takes every three years). Thus, we believe the proposed action is not likely to impede the recovery of, and will not result in an appreciable reduction in, the likelihood of the oceanic whitetip shark's recovery in the wild.

Conclusion

The effects from proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of oceanic whitetip sharks in the wild.

7.6 Giant Manta Ray

The giant manta ray occurs throughout the action area and is likely to be adversely affected by the proposed action; therefore, a jeopardy analysis must determine whether the proposed action will appreciably reduce the likelihood of survival and recovery of this species.

The proposed action may result in 9 non-lethal giant manta ray takes over consecutive 3-year periods. The nonlethal capture of 9 giant manta rays every 3 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover from being captured such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur throughout the action area and captured individuals would be released within the general area where caught, no change in the distribution of this species is anticipated.

There is currently no accurate population estimate for giant manta rays. Giant manta rays can be found worldwide. The best available data indicate that the species has suffered population declines of significant magnitude (up to 95 percent in some places) in the Indo-Pacific and Eastern Pacific portion of its range. NMFS noted that these declines are largely based on trends in landings and market data, diver sightings, and anecdotal observations. The species is not considered threatened in the Atlantic; however, if the species was hypothetically extirpated within the Indo-Pacific and eastern Pacific portion of the range, only the potentially small and fragmented Atlantic populations would remain. The demographic risks associated with small and fragmented populations discussed in the proposed rule, such as demographic stochasticity, dispensation, and inability to adapt to environmental changes, would become significantly greater threats to the species as a whole, and coupled with the species' inherent vulnerability to depletion, indicate that even low levels of mortality would portend drastic declines in the population.

The non-lethal take of 9 giant manta rays over consecutive 3-year periods will not reduce the number of giant manta rays relative to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. This non-lethal take is not expected to result in the loss of reproduction. Therefore, we believe the non-lethal take of 9 giant manta rays over consecutive 3-year periods will not result in population level impacts nor will it change their distribution. Thus, we believe the proposed action is not likely to appreciably reduce the likelihood of survival of the giant manta rays in the wild.

Since giant manta rays were recently listed, a recovery plan for them is not yet available. However, the first step in recovering a species is to reduce identified threats; only by alleviating threats can lasting recovery be achieved. The Final Listing Rule (83 FR 2916, January 22, 2018) noted that overall, current management measures that are in place for fishermen under U.S. jurisdiction appear to directly and indirectly contribute to the infrequency of interactions between U.S. fishing activities and the threatened giant manta ray. As such, NMFS does not believe these activities are contributing significantly to the identified threats of overutilization and inadequate regulatory measures and did not find that developing regulations under section 4(d) to prohibit some or all of these activities is necessary and advisable for the conservation of the species (considering the U.S. interaction with the species is negligible and its moderate risk of extinction is primarily a result of threats from foreign fishing activities). Because the major threat currently contributing to the species' decline is overutilization in waters outside of U.S. jurisdiction, any conservation actions for the giant manta ray that would bring it to the point that the measures of the ESA are no longer necessary will ultimately need to be implemented by foreign nations.

Recovery is the process by which the ecosystems of giant manta rays are restored and the threats to the species are removed. Restoring ecosystems and eliminating threats will support self-populating and self-regulating populations so they can become persistent members of the native biological communities (USFWS and NMFS 1998). As discussed previously, the proposed action is not likely to impede giant manta rays from continuing to survive. The proposed action will not impede the process of restoring the ecosystems that

affect giant manta rays nor are these fisheries a significant threat (9 non-lethal takes every three years). Thus, we believe the proposed action is not likely to impede the recovery of, and will not result in an appreciable reduction in, the likelihood of the giant manta ray's recovery in the wild.

Conclusion

The effects from the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of giant manta rays in the wild.

8.0 Conclusion

NMFS analyzed the best available data, the status of the species, environmental baseline, effects of the proposed action, and cumulative effects to determine whether the proposed action is likely to jeopardize the continued existence of sea turtles, Atlantic sturgeon, smalltooth sawfish, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, the oceanic whitetip shark, or the giant manta ray. Since no critical habitat will be adversely affected the action is not likely to destroy or adversely modify designated critical habitat.

Sea Turtles

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of these species in the wild. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of the Northwest Atlantic DPS of loggerhead, Kemp's ridley, the North and South Atlantic DPSs of green, leatherback, and hawksbill sea turtles.

Smalltooth Sawfish

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of this species in the wild. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of the U.S. DPS of smalltooth sawfish.

Atlantic Sturgeon

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of the 5 DPSs of Atlantic sturgeon in the wild. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of the Gulf of Maine, New York Bight, Chesapeake Bay, South Atlantic, and Carolina DPSs of Atlantic sturgeon.

Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of this DPS in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of the Central and Southwest Atlantic DPS of scalloped hammerhead shark.

Oceanic Whitetip Shark

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of this species in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of the oceanic whitetip shark.

Giant Manta Ray

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of this species in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of the giant manta ray.

9.0 Incidental Take Statement

Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption.

Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. *Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the RPMs and the terms and conditions of the incidental take statement (ITS) of the Opinion.

The take of the Central and Southwest Atlantic DPS of scalloped hammerhead shark, the oceanic whitetip shark, and the giant manta ray by the proposed action is not prohibited, as no Section 4(d) Rules for the species have been promulgated. However, a recent circuit court case held that non-prohibited incidental take must be included in the ITS.⁶⁷ Providing an exemption from Section 9 liability is not the only important purpose of specifying take in an incidental take statement. Specifying incidental take ensures we have a metric against which we can measure whether reinitiation of consultation is required. It also ensures that we identify reasonable and prudent measures we believe are necessary or appropriate to minimize the impact of such incidental take.

⁶⁷ *CBD v. Salazar*, 695 F.3d 893 (9th Cir. 2012). Though the *Salazar* case is not a binding precedent for this action outside of the 9th Circuit, SERO finds the reasoning persuasive and is following the case out of an abundance of caution and anticipation the ruling will be more broadly followed in future cases.

9.1 Anticipated Incidental Take

NMFS anticipates the following incidental takes of sea turtles, smalltooth sawfish, Atlantic sturgeon, Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray may occur in the future because of the proposed action.

The level of takes occurring annually is variable and influenced by sea temperatures, species abundances, and other factors that cannot be predicted. Because of this variability, it is unlikely that all species evaluated in this Opinion will be consistently impacted year after year. For example, some years may have no observed or otherwise documented interactions and thus no estimated take. As a result, monitoring fisheries using 1-year estimated take levels is largely impractical. For these reasons, and based on our experience monitoring fisheries, we believe a 3-year time period is appropriate for meaningful monitoring of take and compliance with the ITS. The triennial takes are set as 3-year running sums (total for any 3-year period) and not for static 3-year periods (i.e., 2019-2021, 2020-2022, 2021-2023 and so on, as opposed to 2018-2020, 2021-2023, 2024-2026, etc.). This approach will allow us to reduce the likelihood of requiring reinitiation unnecessarily because of inherent variability in take levels, but still allow for an accurate assessment of how the proposed action is performing versus our expectations. Table 9.1 displays our 3-year take estimates.

Table 9.1 Anticipated Future 3-Year Take Estimates for the Proposed Action

Sea Turtles	Non-Lethal Take	Lethal Take	Total Estimated Take
Loggerhead	40	51	91
Kemp's ridley	11	11	22
N. Atlantic Green DPS	21	25	46
S. Atlantic Green DPS	1	2	3
Leatherback	3	4	7
Hawksbill	1	1	2
Marine Fish	Non-Lethal Take	Lethal Take	Total Estimated Take
Smalltooth sawfish	22	1	23
Atlantic Sturgeon GOM DPS	26	8	34
Atlantic Sturgeon NYB DPS	134	36	170
Atlantic Sturgeon CB DPS	31	9	40
Atlantic Sturgeon Carolina DPS	5	5	10
Atlantic Sturgeon SA DPS	56	19	75
Atlantic Sturgeon All DPSs	All DPSs = 252	All DPSs = 77	All DPSs = 329
Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS	3	4	7
Oceanic Whitetip Shark	3	3	6
Giant Manta Ray	9	0	9
GOM = Gulf of Maine, NYB = New York Bight, CB = Chesapeake Bay, and SA = South Atlantic			

9.2 Effect of the Take

NMFS has determined the level of anticipated take specified in Section 9.1 is not likely to jeopardize the continued existence of the following ESA-listed species or distinct population segments: Northwest Atlantic DPS of loggerhead sea turtles, South and North Atlantic DPSs of green sea turtles, leatherback sea turtles, Kemp's ridley sea turtles, and hawksbill sea turtles), smalltooth sawfish, Atlantic sturgeon (the Gulf of Maine, Chesapeake, New York Bight, Carolina and the South Atlantic DPSs), scalloped hammerhead shark (Central and Southwest Atlantic DPS), oceanic whitetip shark, and giant manta ray.

9.3 Reasonable and Prudent Measures (RPMs)

Section 7(b)(4) of the ESA requires NMFS, in its role as the consulting agency, to issue to any agency whose proposed action is found to comply with Section 7(a)(2) of the ESA, but may incidentally take individuals of listed species, a statement specifying the impact of that taking. It also states that RPMs necessary to minimize the impacts from the agency action, and terms and conditions to implement those measures, must be provided and followed. Only incidental taking that complies with the specified terms and conditions is authorized.

The RPMs and terms and conditions are required, per 50 CFR 402.14(i)(1)(ii) and (iv), to minimize the impact of the incidental take by the proposed action on ESA-listed species and to ensure compliance with those measures. These measures and terms and conditions are non-discretionary, and must be implemented by NMFS, in its role as the action agency, for the protection of Section 7(o)(2) to apply. If it fails to adhere to the terms and conditions of the ITS through enforceable terms, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse. To monitor the impact of the incidental take, the HMS Management Division must report the progress of the action and its impact on the species to SERO PRD as specified in the ITS [50 CFR 402.14(i)(3)].

We have determined that the following RPMs are necessary or appropriate to minimize the impacts of future takes of sea turtles and ESA-listed fish by the proposed action and to monitor levels of incidental take.

1. Sea Turtle, Smalltooth Sawfish, Atlantic Sturgeon, Scalloped Hammerhead Shark, Oceanic Whitetip Shark, and Giant Manta Ray Handling Requirements

Most, if not all, sea turtles and ESA-listed fish released after entanglement and/or forced submergence events have experienced some degree of physiological injury. The ultimate severity of these events is dependent not only upon actual interaction (i.e., physical trauma from entanglement/forced submergence), but also on the amount of gear remaining on the animal at the time of release. The manner of handling an animal also greatly affects its chance of recovery. Therefore, the experience, ability, and willingness of fishermen to remove gear are crucial to the survival of sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays following release. The HMS Management Division shall ensure that fishermen in the HMS

fisheries that take listed species receive relevant outreach materials describing how captured sea turtles and ESA-listed fish should be handled to minimize adverse effects from incidental take and reduce mortality.

2. Investigate Bottom Longline Soak Time Restrictions

In the shark bottom longline fishery, the gear is set such that the hooked lines are not long enough to reach the surface; therefore, sea turtles are not able to surface and breathe if hooked in this fishery. Carlson et al. (2017) determined the only significant factor affecting sea turtle at-vessel mortality was soak time. The HMS Management Division shall work with the SEFSC and HMS bottom longline fishermen to continue to investigate ways to monitor and limit soak times to minimize sea turtle at-vessel mortalities.

3. Monitoring the Frequency and Magnitude of Incidental Take:

The jeopardy analyses for sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays are based on the assumptions that the frequency and magnitude of anticipated take that occurred in the past will continue into the future. If our estimates regarding the frequency and magnitude of incidental take prove to be an underestimate, we risk having misjudged the potential adverse effects to these species. Thus, it is imperative that we monitor and track the level of take occurring specific to the proposed action. Therefore, the HMS Management Division must ensure that monitoring and reporting of any sea turtle or ESA-listed fish bycatch: (1) detect any adverse effects resulting from the proposed action; (2) assess the actual level of incidental take in comparison with the anticipated incidental take documented in this Opinion; and (3) detect when the level of anticipated take is exceeded.

9.4 Terms and Conditions

To be exempt from take prohibitions established by Section 9 of the ESA, the HMS Management Division must comply with the following terms and conditions, which implement the RPMs described above. These terms and conditions are mandatory.

The following terms and conditions implement RPM No. 1.

1. The HMS Management Division must distribute outreach information to all HMS shark bottom longline and shark gillnet fishermen regarding the sea turtle handling and resuscitation requirements that fishermen must undertake, as stated in 50 CFR 223.206(d)(1) and the NOAA Technical Memorandum NMFS-SEFSC-735: Careful Release Protocols for Sea Turtle Release with Minimal Injury (i.e., NMFS 2019). The HMS Management Division must maintain information on sea turtle release handling and resuscitation requirements and guidelines on its website so that it is accessible to all fishermen, including fishermen using vertical line gear. The HMS Management Division shall annually coordinate with SERO PRD and the SEFSC to check for any updates to the guidance that may need to be distributed.

2. The HMS Management Division must ensure the most recently available [Sawfish Safe Handling and Release Guidance](#) is distributed to all HMS shark bottom longline and shark gillnet fishermen that fish off Florida. HMS Management Division must maintain the smalltooth sawfish safe handling and release guidance on its website so that it is accessible to all fishermen, including fishermen using rod and reel gear. Further, the HMS Management Division shall annually coordinate with SERO PRD to check for any updates to the guidance that may need to be distributed.
3. The HMS Management Division must ensure the most recently available general handling of sturgeon guidance included in the Sturgeon Requirements for Handling Incidentally Taken Sturgeon and Collecting Genetic Samples document (see Appendix B) is distributed to all HMS shark gillnet fishermen that fish off the U.S. East Coast. HMS Management Division must maintain the sturgeon safe handling guidance on its website so that it is accessible to all fishermen. Further, the HMS Management Division shall annually coordinate with SERO PRD to check for any updates to the guidance that may need to be distributed.
4. The HMS Management Division must coordinate with SERO PRD to modify existing safe handling and release guidelines for all shark species to focus on release of oceanic whitetip sharks and Central and Southwest Atlantic DPS of scalloped hammerhead shark within 30-days of issuance of this Opinion. HMS shall ensure that guidance is made available to HMS vertical line fishermen upon completion. HMS Management Division must maintain the oceanic whitetip shark and Central and Southwest Atlantic DPS of scalloped hammerhead shark safe handling and release guidance on its website so that it is accessible to all fishermen, including fishermen using rod and reel, bandit, buoy, and handline gear. Further, the HMS Management Division shall annually coordinate with SERO PRD to check for any updates to the guidance that may need to be distributed. The guidance should include requirements that are at least as protective as the following conditions:
 - a) Leave the shark, especially the gills, in the water as much as possible and remove hook/gear. If the hook cannot be removed, cut the line as close to the hook as possible.
 - b) Do not gaff, or pull the shark by tail, or pick it up by gill slits or spiracles. Do not lift hammerhead sharks by the sides of their heads.
 - c) Release head first. If the shark is a smaller juvenile, and they do not swim off immediately when releasing, move the shark back and forth in the water to aid with water flow through the gills. For hammerheads, you can hold the front of the hammer (cephalofoil) (not the sides) without getting your hand too close to the mouth, and use this to move the body in a figure 8 motion.
5. The HMS Management Division must coordinate with SERO PRD to finalize safe handling and release guidance for giant manta rays within 30-days of issuance of this biological opinion. The guidance must be consistent with and at least as protective as the safe handling and release guidelines in Carlson et al. (2018) at <https://www.fisheries.noaa.gov/species/giant-manta-ray>. HMS shall ensure that guidance is distributed to all HMS shark bottom longline and shark gillnet fishermen

upon completion. HMS Management Division must maintain the giant manta ray safe handling and release guidance on its website so that it is accessible to all fishermen. Further, the HMS Management Division shall annually coordinate with SERO PRD to check for any updates to the guidance that may need to be distributed.

The following terms and conditions implement RPM No. 2

6. The HMS Division must work with the SEFSC to continue investigation of soak time regulations for the HMS shark bottom longline by reviewing SEFSC research on current soak times and catch rates and considering any SEFSC recommended studies or action for further evaluating soak time restrictions arising from those studies. The HMS Division must include information on the status of these investigations in its annual reporting to SERO PRD.

The following terms and conditions implement RPM No. 3.

7. The HMS Management Division must collaborate with the appropriate observer program (e.g., the Northeast Observer Program for smoothhound shark observer data), to ensure the appropriate observer data logs are used to collect data on the HMS-observed fisheries and the appropriate observer data collection protocols are followed.
8. The HMS Management Division must collaborate with the appropriate observer program to ensure that observers are prepared and trained to correctly and safely tag and/or collect samples from incidentally taken sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, and manta rays.
 - a) *Sea Turtles*: For incidentally taken sea turtles, observers must collect tissue samples for genetic analysis. This Opinion serves as the authority for taking associated with observer handling, identifying, measuring, weighing, photographing, flipper tagging, passive integrated transponder (PIT) tagging, skin biopsy and releasing incidentally taken sea turtles (without the need for an ESA Section 10 permit). Samples collected must be analyzed to determine the genetic identity of individual sea turtles caught
 - b) *Smalltooth Sawfish*: For incidentally taken smalltooth sawfish, observers must be trained to tag smalltooth sawfish. Sampling protocols for observers will be provided by Protected Resources Division and include such details as what measurements to take, how to properly take measurements, what samples to collect, how to store samples, etc. The HMS Management Division shall annually coordinate with the Protected Resources Division to check for any updates to the protocols that may need to be distributed. All dead carcasses of smalltooth sawfish must be placed on ice and transferred to the SEFSC, attention Dr. John Carlson (National Marine Fisheries Service, Panama City Laboratory, 3500 Delwood Beach Rd, Panama City, FL, 32408).
 - c) *Atlantic Sturgeon*: For incidentally taken Atlantic sturgeon, observers must be trained to tag them, take a tissue sample, and scan them for PIT tags. Observers must collect a tissue sample from any Atlantic sturgeon handled onboard an HMS

fishing vessel. A total length measurement or estimate, time and location (i.e., lat./long. and approximate water depth) of capture, circumstances of capture, and status (i.e., dead, alive, injured) upon return to the water should accompany the tissue sample. Observers must follow the Sturgeon Requirements for Handling Incidentally Taken Sturgeon and Collecting Genetic Samples (see Appendix B).

- d) *Listed Sharks and Rays*: For incidentally taken scalloped hammerhead sharks, oceanic whitetip sharks, and/or giant manta rays, observers must be trained to properly collect necessary scientific data. The HMS Management Division must work with the SEFSC and SERO PRD to modify existing sampling protocols for oceanic whitetip shark, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and giant manta ray, within 30-days of issuance of this Opinion. HMS Management Division shall ensure that guidance is distributed to observers upon completion. Further, the HMS Management Division shall annually coordinate with the SERO PRD to check for any updates to the protocols that may need to be distributed. For example, the protocols should consider the following:

Length measurements/disc widths, sex (if discernible), time and location (i.e., lat./long. and approximate water depth) of capture, any identifying marks or tags, and status (i.e., dead, alive, injured) at landing and upon return to the water should be reported. Observers must record the information as specified on the observer form. These forms should be submitted in accordance with Term and Condition No. 11, below.

9. HMS Management Division, in collaboration with the NEFSC/GARFO and SEFSC/SERO, must maintain the standardized protocol developed following the 2012 Opinion for determining which trips, and how much effort, were directed toward smoothhound. Determining directed fishing effort levels in the smoothhound fishery and avoiding double reporting or underreporting of effort, as well as identifying any effort shifts that may occur, is necessary to monitor incidental takes of ESA-listed species in directed smoothhound fishing.
10. The HMS Management Division must continue to work with the appropriate observer program (i.e., NEFSC and SEFSC observer programs) to ensure observer coverage in observed HMS fisheries subject to this consultation is sufficient for monitoring take of ESA-listed species. NMFS (2004d) recommends a level of observer coverage equal to that which provides estimates of a protected species interaction with an expected coefficient of variation of 30%. Since ESA-listed species are relatively rare, achieving bycatch estimates with CVs of 30% or less may not be feasible. If the HMS Management Division, in conjunction with the appropriate observer program, determines achieving CVs less than 30% are not possible, NMFS must provide information on the observer coverage and bycatch estimates, including the CVs around the bycatch estimates, and explain why those bycatch estimates are the best scientific data available to monitor take. NMFS must note any changes to observer coverage, and any resulting changes to CVs for the bycatch estimates from prior years.

11. The HMS Management Division, in collaboration with the appropriate Science Center (i.e., NEFSC, SEFSC) must (1) collect and monitor observer and other reports (i.e., reports from MRIP) from HMS targeted trips having sea turtle, smalltooth sawfish, Atlantic sturgeon, Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark, or giant manta ray interactions and (2) submit an annual report detailing these interactions to SERO PRD; the information below must also be included. The required information may be included in a single report or multiple reports.

a) Information Required for Species Interactions:

- i) *Sea Turtle Reports*: must include all information specified on the SEFSC sea turtle life history form for any sea turtle captured.
- ii) *Smalltooth Sawfish Reports*: must include a length measurement or estimate, time and location (i.e., lat./long. and approximate water depth) of capture, circumstances of capture (e.g., position of sawfish in the trawl net), and status (i.e., condition, sex, alive, injured) upon return to the water must be reported to the extent possible.
- iii) *Atlantic Sturgeon Reports*: must include a total length measurement or estimate, weight measurement or estimate, sex (if discernible), time and location (i.e., lat./long. and approximate water depth) of capture, information whether the fish was tagged and if so what type of tag was used, and status (i.e., dead, alive, injured) upon return to the water should be reported.
- iv) *Shark Reports*: for scalloped hammerhead sharks and oceanic whitetip sharks, observers must include a length measurement or estimate, weight measurement or estimate, sex (if discernible), time and location (i.e., lat./long. and approximate water depth) of capture, information on whether the shark was tagged, and if so what type of tag was used, and status (i.e., dead, alive, injured) upon return to the water should be reported.
- v) *Manta Reports*: must include a disk width (DW) measurement or estimate (i.e., DW is a straight line measurement from wing tip to wing tip), time and location (i.e., lat./long. and approximate water depth) of capture, and status (i.e., dead, alive, injured) upon return to the water should be reported.

b) Information Required on Fishery Operations

- i) *Gillnet Gear*: type of gear used (e.g., drift, sink, strike), set date, net length (ft), net depth (ft), minimum stretched mesh size (in), soak time (hrs), trip length, number of sets per trip, whether tie-downs were used, and length of tie-down if used.
- ii) *Bottom Longline Gear*: mainline length (ft), depth fished (ft), number of sets, number of lines per set, number of hooks fished per set, hook type (e.g., size of circle and any offset), soak time (hrs), and bait used.

c) Reports must also estimate the total rolling three year take in HMS fisheries subject to this consultation based on availability of effort data and reported and observed

takes. If the estimated take of sea turtles, smalltooth sawfish, Atlantic sturgeon, scalloped hammerhead sharks, oceanic whitetip sharks, or giant manta rays, is higher than anticipated in this Opinion, the report should include an analysis of the possible reasons for the higher than expected level of take and whether this higher level of take is expected to occur again.

- d) These reports must be forwarded to the NMFS Assistant Regional Administrator for Protected Resources, Southeast Regional Office, Protected Resources Division, 263 13th Avenue South, St. Petersburg, Florida 33701-5505.

10.0 Conservation Recommendations

Section 7(a)(1) of the ESA directs federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities 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.

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The following additional measures are recommended for listed species adversely affected by the proposed action and for North Atlantic right whales. Although the proposed action is not likely to adversely affect North Atlantic right whales, the Southeast gillnet fishery analyzed in this Opinion has been found to adversely affect this species in the past consultations. Our determination in this Opinion is based on regulations implemented under the ALWTRP reducing risk of entanglement in shark gillnets to discountable levels. Consequently, we think conservation recommendations for North Atlantic right whale protection are still relevant to the proposed action. For SERO PRD to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, SERO PRD requests notification of the implementation of any conservation recommendations.

North Atlantic Right Whales:

1. 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's 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.
2. NMFS should continue to undertake and support aerial surveys, passive acoustic monitoring, and the Sighting Advisory System.
3. NMFS should continue to develop and implement measures to reduce the risk of ship strikes of large whales.
4. 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.
5. NMFS should continue to cooperate with the Canadian government to compare research findings and facilitate implementation in both countries of the most promising risk reduction practices for large whales.

6. In general, NMFS should avoid allowing any new fishing activities (e.g., exempted fishing permits) during the November through April timeframe within North Atlantic right whale critical habitat.

Sea Turtles:

7. NMFS should support in-water abundance estimates of sea turtles to achieve more accurate status assessments for these species and to better assess the impacts of incidental take during HMS fishing.
8. Once reasonable in-water estimates are obtained, NMFS should support population modeling or other risk analyses of the sea turtle populations affected by the HMS fisheries. This will help improve the accuracy of future assessments of the effects of different levels of take on sea turtle populations.

Smalltooth Sawfish:

9. NMFS should conduct or fund research or alternative methods (e.g., surveys) on the distribution, abundance, and migratory behavior of smalltooth sawfish in federal fishing areas off south Florida and the Florida Keys, to better understand their occurrence in federal waters and potential for interaction with HMS fisheries.
10. NMFS should conduct or fund reproductive studies to ensure that any incidental capture of smalltooth sawfish during fishing activities is not disrupting any such activities.

Atlantic Sturgeons:

11. NMFS should fund or conduct future research to identify migration patterns of ESA-listed sturgeon. Telemetry studies to track fish and ascertain the use of spawning and foraging habitat would improve knowledge of life history. Data describing the upstream sturgeon spawning areas to characterize habitat and assess availability would assist in determining spawning habitat preference and availability.
12. NMFS should fund or conduct future research that evaluates the relationship between flow, water temperature, and sturgeon migration. Additional information on this relationship would provide a better indicator of conditions that cue and successfully initiate sturgeon spawning movement.
13. NMFS should collect data describing Atlantic sturgeon location and movement in the Atlantic Ocean, by depth and substrate to assist in future assessments of interactions between fishing gear (i.e., commercial, recreational, or research) sturgeon migratory and feeding behavior.
14. NMFS should collect information on incidental catch rates and condition of sturgeon captured in HMS gear to assist in future assessments of gear impacts to sturgeon.

Sharks:

15. Given the ESA listings for Central and Southwest Atlantic DPS of scalloped hammerhead and oceanic whitetip, SERO PRD strongly encourages the HMS Management Division to include these federally protected species on the HMS list of Prohibited Shark species for recreational and/or commercial HMS fisheries.

This effort would promote conservation and recovery of these threatened species. While retention and possession of oceanic whitetip and scalloped hammerhead sharks are already prohibited in the PLL fishery, consistent with regulations implementing various ICCAT recommendations, this prohibition does not extend to all HMS fisheries. Therefore, further protections are warranted.

16. NMFS should support research investigating the location of scalloped hammerhead nursery areas for both listed and non-listed DPSs.
17. NMFS should support research that investigates ways to reduce and minimize at-vessel mortality of scalloped hammerhead sharks and oceanic whitetips sharks in commercial and recreational fisheries.
18. NMFS should conduct research on gear modifications to increase survivorship of oceanic whitetip sharks when caught in commercial fisheries.
19. NMFS should expand existing research of at-vessel and post-release mortality rates of oceanic whitetip sharks in longlines and purse seines to improve stock assessments.
20. NMFS should conduct surveys of fishermen regarding the effectiveness of safe release techniques for oceanic whitetip sharks
21. NMFS should continue research on bycatch mitigation measures to minimize interactions in gillnet and longline fisheries and share best practices (knowledge/technology transfer)
22. NMFS should conduct research to better estimate oceanic whitetip shark post-release mortality.

Giant Manta Rays:

23. NMFS should continue to develop guidance for fishing practices that minimize bycatch, including handling and release procedures using different gears, and produce education and outreach materials about safe handling and release.
24. NMFS should collect data or fund research to estimate post-release mortality across various sizes and gear types.
25. NMFS should conduct or fund research that describe and define areas of critical habitat and population connectivity (by size, sex and reproductive status), including areas of core use (aggregation and foraging sites), seasonality of presence, and movement/migratory corridors.
26. NMFS should conduct or fund research that estimates abundance of using information collected by fisheries-independent research programs (e.g., line transect surveys, photo identification, tagging).
27. NMFS should create education and outreach material to communicate manta ray conservation messages through social media, websites, magazines, and print to federal agencies, local communities, and non-governmental organizations (NGOs).

11.0 Reinitiation of Consultation

This concludes formal consultation on the proposed action. As provided in 50 CFR 402.16, reinitiation of formal consultation is required if discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if (1) the amount or extent of the taking specified in the incidental take statement is exceeded; (2) new information reveals effects of the agency action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered in this Opinion; (3) the identified action is subsequently modified in a manner that causes an effect to listed species or critical habitat that was not considered in the Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the HMS Management Division must immediately request reinitiation of formal consultation.

12.0 Literature Cited

- ABRT, and coauthors. 2005. Atlantic *Acropora* status review document. Acropora Biological Review Team, National Marine Fisheries Service, Southeast Regional Office. .
- Ackerman, R. A. 1997. The nest environment and the embryonic development of sea turtles. Pages 83-106 in P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.
- Adams, D. H., and R. Paperno. 2007. Preliminary Assessment of a Neashore Nursery Ground for the Scalloped Hammerhead off the Atlantic Coast of Florida. Pages 165 in American Fisheries Society Symposium. American Fisheries Society.
- Addison, D. 1997. Sea turtle nesting on Cay Sal, Bahamas, recorded June 2-4, 1996. Bahamas Journal of Science 5(1):34-35.
- Addison, D., and B. Morford. 1996. Sea turtle nesting activity on the Cay Sal Bank, Bahamas. Bahamas Journal of Science 3(3):31-36.
- Aguilar, R., J. Mas, and X. Pastor. 1994. Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle *Caretta caretta* population in the western Mediterranean. Pages 91-96 in J. I. Richardson, and T. H. Richardson, editors. Proceedings of the 12th Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Jekyll Island, Georgia.
- Aguilar, R., J. Mas, and X. Pastor. 1995. Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle *Caretta caretta* population in the western Mediterranean. J. I. Richardson, and T. H. Richardson, editors. Twelfth Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Jekyll Island, Georgia.
- Aguirre, A., G. Balazs, T. Spraker, S. K. K. Murakawa, and B. Zimmerman. 2002. Pathology of oropharyngeal fibropapillomatosis in green turtles *Chelonia mydas*. Journal of Aquatic Animal Health 14:298-304.
- Aguirre, A. A., G. H. Balazs, B. Zimmerman, and F. D. Galey. 1994. Organic Contaminants and Trace Metals in the Tissues of Green Turtles (*Chelonia mydas*) Afflicted with Fibropapillomas in the Hawaiian Islands. Marine Pollution Bulletin 28(2):109-114.
- Amorim, A. F., C. A. Arfelli, and L. Fagundes. 1998. Pelagic elasmobranchs caught by longliners off southern Brazil during 1974-97: An Overview. Marine and Freshwater Research 49(7):621-632.

- Amos, A. F. 1989. The occurrence of Hawksbills (*Eretmochelys imbricata*) along the Texas Coast. Pages 9-11 in S. A. Eckert, K. L. Eckert, and T. H. Richardson, editors. Ninth Annual Workshop on Sea Turtle Conservation and Biology.
- Antonelis, G. A., J. D. Baker, T. C. Johanos, R. C. Braun, and A. L. Harting. 2006. Hawaiian monk seal (*Monachus schauinslandi*): Status and conservation issues. Atoll Research Bulletin 543:75-101.
- Arendt, M., and coauthors. 2009. Examination of local movement and migratory behavior of sea turtles during spring and summer along the Atlantic coast off the southeastern United States. South Carolina Department of Natural Resources, Marine Resources Division.
- Armstrong, J. L., and J. E. Hightower. 2002. Potential for restoration of the Roanoke River population of Atlantic sturgeon. Journal of Applied Ichthyology 18(4-6):475-480
- ASMFC. 1998. American shad and Atlantic sturgeon stock assessment peer review: Terms of reference and advisory report. Atlantic States Marine Fisheries Commission, Washington, D. C.
- ASMFC. 2007. Special Report to the Atlantic Sturgeon Management Board: Estimation of Atlantic sturgeon bycatch in coastal Atlantic commercial fisheries of New England and the Mid-Atlantic. Atlantic States Marine Fisheries Commission, Arlington, Virginia.
- ASMFC. 2010. Atlantic States Marine Fisheries Commission Annual Report.
- ASMFC. 2017. Atlantic Sturgeon Benchmark Stock Assessment and Peer Review Report. Atlantic States Marine Fisheries Commission, Arlington, VA.
- ASSRT. 1998. Status Review of Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Atlantic Sturgeon Status Review Team, Gloucester, Massachusetts.
- ASSRT. 2007. Status Review of Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Atlantic Sturgeon Status Review Team, Gloucester, Massachusetts.
- Avens, L., J. C. Taylor, L. R. Goshe, T. T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8(3):165-177.
- Backus, R. H., S. Springer, and E. L. Arnold. 1956. A contribution to the natural history of the white-tip shark, *Pterolamiops longimanus* (Poey). Deep Sea Research (1953) 3(3):178-188.

- Bahr, D. L., and D. L. Peterson. 2016. Recruitment of juvenile Atlantic sturgeon in the Savannah River, Georgia Transactions of the American Fisheries Society 145(6):1171-1178.
- Bain, M., N. Haley, D. Peterson, J. R. Waldman, and K. Arend. 2000. Harvest and habitats of Atlantic sturgeon *Acipenser oxyrinchus* Mitchill, 1815 in the Hudson River estuary: lessons for sturgeon conservation. Boletín. Instituto Español de Oceanografía 16:43-53.
- Bain, M. B. 1997. Atlantic and shortnose sturgeons of the Hudson River: Common and divergent life history attributes. Environmental Biology of Fishes 48(1-4):347-358.
- Baker, J., C. Littnan, and D. Johnston. 2006. Potential effects of sea-level rise on terrestrial habitat and biota of the northwestern Hawaiian Islands. Pages 3 in Twentieth Annual Meeting Society for Conservation Biology Conference, San Jose, California.
- Balazik, M. T., D. J. Farrae, T. L. Darden, and G. C. Garman. 2017. Genetic differentiation of spring-spawning and fall-spawning male Atlantic sturgeon in the James River, Virginia. PLoS ONE 12(7):e0179661.
- Balazik, M. T., G. C. Garman, J. P. Van Eenennaam, J. Mohler, and L. C. Woods. 2012a. Empirical Evidence of Fall Spawning by Atlantic Sturgeon in the James River, Virginia. Transactions of the American Fisheries Society 141(6):1465-1471.
- Balazik, M. T., and J. A. Musick. 2015. Dual annual spawning races in Atlantic Sturgeon. PLoS One 10(5):e0128234.
- Balazik, M. T., and coauthors. 2012b. The potential for vessel interactions with adult Atlantic sturgeon in the James River, Virginia. North American Journal of Fisheries Management 32(6):1062-1069.
- Balazs, G. H. 1982. Growth rates of immature green turtles in the Hawaiian Archipelago. Pages 117-125 in K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C.
- Balazs, G. H. 1983. Recovery records of adult green turtles observed or originally tagged at French Frigate Shoals, Northwestern Hawaiian Islands. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, NOAA-TM-NMFS-SWFC-36.
- Balazs, G. H. 1985a. Impact of ocean debris on marine turtles: Entanglement and ingestion Pages 387-429 in R. S. Shomura, and H. O. Yoshida, editors. Workshop on the Fate and Impact of Marine Debris, Honolulu, Hawaii.

- Balazs, G. H. 1985b. Sea turtles and debris: Ingestion and entanglement. *Marine Turtle Newsletter*:8-9.
- Balazs, G. H., S. G. Pooley, and S. K. Murakawa. 1995. Guidelines for handling marine turtles hooked or entangled in the Hawaii longline fishery: Results of an expert workshop held in Honolulu, Hawaii March 15-17,1995. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Honolulu.
- Bardach, J. E. 1958. On the movements of certain Bermuda reef fishes. *Ecology* 39(1):139-146.
- Bardach, J. E., C. L. Smith, and D. W. Menzel. 1958. Bermuda fisheries research program final report. Bermuda Trade Development Board, Hamilton, Bermuda.
- Baremore, I. E., J. K. Carlson, L. D. Hollensead, and D. M. Bethea. 2007. Catch and Bycatch in U.S. Southeast Gillnet Fisheries, 2007. NOAA Technical Memorandum NMFS-SEFSC-565.
- Barnette, M. C. 2001. A review of the fishing gear utilized within the Southeast Region and their potential impacts on essential fish habitat. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.
- Bass, A. L., and coauthors. 1996. Testing models of female reproductive migratory behaviour and population structure in the Caribbean hawksbill turtle, *Eretmochelys imbricata*, with mtDNA sequences. *Molecular Ecology* 5:321-328.
- Bass, A. L., C. J. Lagueux, and B. W. Bowen. 1998. Origin of green turtles, *Chelonia mydas*, at "Sleeping Rocks" off the northeast coast of Nicaragua. *Copeia* 1998(4):1064-1069.
- Bass, A. L., and W. N. Witzell. 2000. Demographic composition of immature green turtles (*Chelonia mydas*) from the east central Florida coast: Evidence from mtDNA markers. *Herpetologica* 56(3):357-367.
- Baughman, J. L. 1943. Notes on Sawfish, *Pristis perotteti* Müller and Henle, not previously reported from the waters of the United States. *Copeia* 1943(1):43-48.
- Baum, J., E. Medina, J. A. Musick, and M. Smale. 2006. *Carcharhinus longimanus*. 2011 IUCN Red List of Threatened Species. International Union for Conservation of Nature and Natural Resources.
- Baum, J. K., and coauthors. 2003. Collapse and conservation of shark populations in the northwest Atlantic. *Science* 299:389-392.
- Beauvais, S. L., S. B. Jones, S. K. Brewer, and E. E. Little. 2000. Physiological measures of neurotoxicity of diazinon and malathion to larval rainbow trout (*Oncorhynchus*

mykiss) and their correlation with behavioral measures. *Environmental Toxicology and Chemistry* 19(7):1875-1880.

- Beerkircher, L. R., E. Cortes, and M. Shivji. 2002. Characteristics of shark bycatch observed on pelagic longlines off the southeastern United States, 1992–2000. *Marine Fisheries Review* 64(4):40-49.
- Bejarano-Álvarez, M., F. Galvan-Magana, and R. I. Ochoa-Baez. 2011. Reproductive biology of the scalloped hammerhead shark *Sphyrna lewini* (Chondrichthyes: Sphyrnidae) off south-west Mexico. *International Journal of Ichthyology* 17(1):11-23.
- Belovsky, G. E. 1987. Extinction models and mammalian persistence. Chapter 3 *In*: Soulé, M.E. (ed), *Viable Populations for Conservation*. Cambridge University Press, pp.35-57.
- Benson, S. R., and coauthors. 2007a. Post-nesting migrations of leatherback turtles (*Dermochelys coriacea*) from Jamursba-Medi, Bird's Head Peninsula, Indonesia. *Chelonian Conservation and Biology* 6(1):150-154.
- Benson, S. R., and coauthors. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. *Ecosphere* 2(7).
- Benson, S. R., K. A. Forney, J. T. Harvey, J. V. Carretta, and P. H. Dutton. 2007b. Abundance, distribution, and habitat of leatherback turtles (*Dermochelys coriacea*) off California, 1990–2003. *Fishery Bulletin* 105(3):337-347.
- Berkeley, S. A., and W. L. Campos. 1988. Relative abundance and fishery potential of pelagic sharks along Florida's east coast. *Mar. Fish. Rev* 50(1):9-16.
- Berlin, W. H., R. J. Hesselberg, and M. J. Mac. 1981. Chlorinated hydrocarbons as a factor in the reproduction and survival of Lake Trout (*Salvelinus namaycush*) in Lake Michigan. Technical Paper 105. U.S. Fish and Wildlife Service.
- Berry, R. J. 1971. Conservation aspects of the genetical constitution of populations. Pages 177-206 *in* E. D. Duffey, and A. S. Watt, editors. *The Scientific Management of Animal and Plant Communities for Conservation*, Blackwell, Oxford.
- Bessudo, S., and coauthors. 2011. Residency of the scalloped hammerhead shark (*Sphyrna lewini*) at Malpelo Island and evidence of migration to other islands in the Eastern Tropical Pacific. *Environmental Biology of Fishes* 91(2):165-176.
- Bigelow, H. B., and W. C. Schroeder. 1953a. *Fishes of the Gulf of Maine*, volume 53. US Government Printing Office Washington, DC.
- Bigelow, H. B., and W. C. Schroeder. 1953b. *Sawfishes, guitarfishes, skates, and rays*. J. Tee-Van, C. M. Breder, A. E. Parr, W. C. Schroeder, and L. P. Schultz, editors.

Fishes of the Western North Atlantic, Part Two. Sears Foundation for Marine Research, New Haven, CT.

- Billsson, K., L. Westerlund, M. Tysklind, and P.-e. Olsson. 1998. Developmental disturbances caused by polychlorinated biphenyls in zebrafish (*Brachydanio rerio*). *Marine Environmental Research* 46(1–5):461–464.
- Bjorndal, K. A. 1982. The consequences of herbivory for life history pattern of the Caribbean green turtle, *Chelonia mydas*. Pages 111–116 in *Biology and Conservation of Sea Turtles*. Smithsonian Institution, Washington, D. C.
- Bjorndal, K. A. 1997. Foraging ecology and nutrition of sea turtles. Pages 199–231 in *The Biology of Sea Turtles*. CRC Press, Boca Raton, Florida.
- Bjorndal, K. A., and A. B. Bolten. 2002. Proceedings of a workshop on assessing abundance and trends for in-water sea turtle populations. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, NMFS-SEFSC-445.
- Bjorndal, K. A., and A. B. Bolten. 2008. Annual variation in source contributions to a mixed stock: Implications for quantifying connectivity. *Molecular Ecology* 17(9):2185–2193.
- Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. 2005. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. *Ecological Applications* 15(1):304–314.
- Bjorndal, K. A., A. B. Bolten, T. Dellinger, C. Delgado, and H. R. Martins. 2003. Compensatory growth in oceanic loggerhead sea turtles: Response to a stochastic environment. *Ecology* 84(5):1237–1249.
- Bjorndal, K. A., J. A. Wetherall, A. B. Bolten, and J. A. Mortimer. 1999. Twenty-six years of green turtle nesting at Tortuguero, Costa-Rica: An encouraging trend. *Conservation Biology* 13(1):126–134.
- Bolker, B. M., T. Okuyama, K. A. Bjorndal, and A. B. Bolten. 2007. Incorporating multiple mixed stocks in mixed stock analysis: 'many-to-many' analyses. *Molecular Ecology* 16(4):685–695.
- Bolten, A., and B. Witherington. 2003. *Loggerhead Sea Turtles*. Smithsonian Books, Washington, D. C.
- Bolten, A. B., K. A. Bjorndal, and H. R. Martins. 1994. Life history model for the loggerhead sea turtle (*Caretta caretta*) populations in the Atlantic: Potential impacts of a longline fishery. Pages 48–55 in G. J. Balazs, and S. G. Pooley, editors. *Research Plan to Assess Marine Turtle Hooking Mortality*, volume Technical Memorandum NMFS-SEFSC-201. National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

- Bolten, A. B., and coauthors. 1998. Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis. *Ecological Applications* 8(1):1-7.
- Bonfil, R. 2009. The biology and ecology of the silky shark, *Carcharhinus falciformis*. Pages 114-127 in T. J. Pitcher, editor. *Sharks of the Open Ocean*. Blackwell Publishing Ltd., Oxford, UK.
- Bonfil, R., and coauthors. 2008. The biology and ecology of the oceanic whitetip shark, *Carcharhinus longimanus*. *Sharks of the Open Ocean: Biology, Fisheries, and Conservation*:128-139.
- Boreman, J. 1997a. Sensitivity of North American sturgeons and paddlefish to fishing mortality. *Environmental Biology of Fishes* 48(1-4):399-405.
- Boreman, J. 1997b. Sensitivity of North American sturgeons and paddlefish to fishing mortality. *Environmental Biology of Fishes* 48(1):399-405.
- Borodin, N. 1925. Biological Observations on the Atlantic Sturgeon (*Acipenser sturio*). *Transactions of the American Fisheries Society* 55(1):184-190.
- Bostrom, B. L., and D. R. Jones. 2007. Exercise warms adult leatherback turtles. *Comparative Biochemistry and Physiology A: Molecular and Integrated Physiology* 147(2):323-31.
- Bouchard, S., and coauthors. 1998. Effects of exposed pilings on sea turtle nesting activity at Melbourne Beach, Florida. *Journal of Coastal Research* 14(4):1343-1347.
- Boulon, R. H., Jr. 1983. Some notes on the population biology of green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtles in the northern U.S. Virgin Islands: 1981-1983. Report to the National Marine Fisheries Service, Grant No. NA82-GA-A-00044.
- Boulon Jr., R. H. 1994. Growth rates of wild juvenile hawksbill turtles, *Eretmochelys imbricata*, in St. Thomas, United States Virgin Islands. *Copeia* 1994(3):811-814.
- Bowen, B. W., and coauthors. 1992. Global population structure and natural history of the green turtle (*Chelonia mydas*) in terms of matriarchal phylogeny. *Evolution* 46(4):865-881.
- Bowen, B. W., and W. N. Witzell. 1996. Proceedings of the International Symposium on Sea Turtle Conservation Genetics. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, NMFS-SEFSC-396.

- Bowlby, C. E., G. A. Green, and M. L. Bonnell. 1994. Observations of leatherback turtles offshore of Washington and Oregon. *Northwestern Naturalist* 75(1):33-35.
- Branstetter, S. 1987. Age, growth and reproductive biology of the silky shark, *Carcharhinus falciformis*, and the scalloped hammerhead, *Sphyrna lewini*, from the northwestern Gulf of Mexico. *Environmental Biology of Fishes* 19(3):161-173.
- Branstetter, S. 1990. Early life-history implications of selected carcharhinoid and lamnoid sharks of the northwest Atlantic. NOAA Technical Report NMFS 90:17-28.
- Brautigam, A., and K. L. Eckert. 2006. Turning the tide: Exploitation, trade and management of marine turtles in the Lesser Antilles, Central America, Columbia and Venezuela. TRAFFIC International, Cambridge, United Kingdom.
- Bresette, M., R. A. Scarpino, D. A. Singewald, and E. P. de Maye. 2006. Recruitment of post-pelagic green turtles (*Chelonia mydas*) to nearshore reefs on Florida's southeast coast. Pages 288 in M. Frick, A. Panagopoulou, A. F. Rees, and K. Williams, editors. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.
- Broderick, A. C., and coauthors. 2006. Are green turtles globally endangered? *Global Ecology and Biogeography* 15(1):21-26.
- Brundage, H. M., and J. C. O. Herron. 2003. Population estimate for shortnose sturgeon in the Delaware River. Presented at the 2003 Shortnose Sturgeon Conference, 7-9 July 2003.
- Burchfield, P. M. 2013. Gladys Porter Zoo's Preliminary Annual Report on the Mexico/United States of America Population Restoration Project for the Kemp's Ridley Sea Turtle, (*Lepidochelys kempii*), on the Coasts of Tamaulipas, Mexico. Secretaria de Medio Ambiente y Recursos Naturales, Subsecretaria de Gestion Para La Proteccion Ambiental, Direccion General de Vida Silvestre, U.S. Fish and Wildlife Service, Texas Parks and Wildlife Department.
- Bush, A. 2003. Diet and diel feeding periodicity of juvenile scalloped hammerhead sharks, *Sphyrna lewini*, in Kāne'ohe Bay, Ō'ahu, Hawai'i. *Environmental Biology of Fishes* 67(1):1-11.
- Bushnoe, T., J. Musick, and D. Ha. 2005. Essential spawning and nursery habitat of Atlantic sturgeon (*Acipenser oxyrinchus*) in Virginia. Virginia Institute of Marine Science, Gloucester Point, Virginia.
- Caldwell, D. K., and A. Carr. 1957. Status of the sea turtle fishery in Florida. Pages 457-463 in J. B. Trefethen, editor Twenty-Second North American Wildlife Conference. Wildlife Management Institute, Statler Hotel, Washington, D. C.
- Caldwell, S. 1990. Texas sawfish: Which way did they go? *Tide* Jan.-Feb.:16-19.

- Cameron, P., J. Berg, V. Dethlefsen, and H. Von Westernhagen. 1992. Developmental defects in pelagic embryos of several flatfish species in the Southern North sea. *Netherlands Journal of Sea Research* 29(1-3):239-256.
- Campell, C. L., and C. J. Lagueux. 2005. Survival probability estimates for large juvenile and adult green turtles (*Chelonia mydas*) exposed to an artisanal marine turtle fishery in the western Caribbean. *Herpetologica* 61(2):91-103.
- Carballo, J. L., C. Olabarria, and T. G. Osuna. 2002. Analysis of four macroalgal assemblages along the Pacific Mexican coast during and after the 1997-98 El Niño. *Ecosystems* 5(8):749-760.
- Carillo, E., G. J. W. Webb, and S. C. Manolis. 1999. Hawksbill turtles (*Eretmochelys imbricata*) in Cuba: an assessment of the historical harvest and its impacts. *Chelonian Conservation and Biology* 3(2):264-280.
- Carlson, J., and D. M. Bethea. 2007. Catch and Bycatch in the shark gillnet fishery: 2005-2006. NOAA Technical Memorandum NMFS-SEFSC-552, 26p.
- Carlson, J., A. Mathers, and L. W. Stokes. 2017. Protected Species Takes in the Shark Bottom Longline and Shark Gillnet Fishery, 2017. NMFS-SEFSC SFD Contribution PCB-17-03. March 2017.
- Carlson, J. K., L. F. Hale, A. Morgan, and G. Burgess. 2012. Relative abundance and size of coastal sharks derived from commercial shark longline catch and effort data. *Journal of Fish Biology* 80(5):1749-1764.
- Carlson, J. K., and J. Osborne. 2012. Relative abundance of smalltooth sawfish (*Pristis pectinata*) based on the Everglades National Park Creel Survey. NOAA Technical Memorandum NMFS-SEFSC-626. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Panama City Laboratory, NOAA Technical Memorandum NMFS-SEFSC-626, Panama City, Florida.
- Carlson, J. K., J. Osborne, and T. W. Schmidt. 2007. Monitoring the recovery of smalltooth sawfish, *Pristis pectinata*, using standardized relative indices of abundance. *Biological Conservation* 136(2):195-202.
- Carlson, J. K., and P. M. Richards. 2011. Takes of Protected Species in the Northwest Atlantic Ocean and Gulf of Mexico Shark Bottom Longline and Gillnet Fishery 2007-2010.
- Caron, F., D. Hatin, and R. Fortin. 2002. Biological characteristics of adult Atlantic sturgeon (*Acipenser oxyrinchus*) in the St Lawrence River estuary and the effectiveness of management rules. *Journal of Applied Ichthyology* 18(4-6):580-585.

- Carr, A. F. 1986. New perspectives on the pelagic stage of sea turtle development. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center.
- Carr, T., and N. Carr. 1991. Surveys of the sea turtles of Angola. *Biological Conservation* 58(1):19-29.
- Carter, J., G. J. Marrow, and V. Pryor. 1994. Aspects of the ecology and reproduction of Nassau grouper, *Epinephelus striatus*, off the coast of Belize, Central America. Pages 65-111 *in* Proceedings of the Gulf and Caribbean Fisheries Institute. Gulf and Caribbean Fisheries Institute.
- Catry, P., and coauthors. 2002. First census of the green turtle at Poilifilo, Bijagos Archipelago, Guinea-Bissau: The most important nesting colony on the Atlantic coast of Africa. *Oryx* 36(4):400-403.
- Catry, P., and coauthors. 2009. Status, ecology, and conservation of sea turtles in Guinea-Bissau. *Chelonian Conservation and Biology* 8(2):150-160.
- Caurant, F., P. Bustamante, M. Bordes, and P. Miramand. 1999. Bioaccumulation of cadmium, copper and zinc in some tissues of three species of marine turtles stranded along the French Atlantic coasts. *Marine Pollution Bulletin* 38(12):1085-1091.
- Cech, J. J., Jr., and S. I. Doroshov. 2005. Environmental requirements, preferences, and tolerance limits of North American sturgeons. Pages 73-86 *in* Sturgeons and Paddlefish of North America. Springer.
- Cervigón, F. 1994. Los Peces Marinas de Venezuela, volume I and II. Fund. La Salle Ciencia Naturales.
- CETAP. 1982. A characterization of marine mammals and turtles in the mid- and north-Atlantic areas of the U.S. Outer Continental Shelf. Cetacean and Turtle Assessment Program, Bureau of Land Management, BLM/YL/TR-82/03, Washington, D. C.
- Chacón-Chaverri, D., and K. L. Eckert. 2007. Leatherback sea turtle nesting at Gandoca Beach in Caribbean Costa Rica: Management recommendations from fifteen years of conservation. *Chelonian Conservation and Biology* 6(1):101-110.
- Chaloupka, M. 2002. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. *Ecological Modelling* 148(1):79-109.
- Chaloupka, M., and C. Limpus. 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. *Marine Biology* 146(6):1251-1261.
- Chaloupka, M., C. Limpus, and J. Miller. 2004. Green turtle somatic growth dynamics in a spatially disjunct Great Barrier Reef metapopulation. *Coral Reefs* 23(3):325-335.

- Chaloupka, M., T. M. Work, G. H. Balazs, S. K. K. Murakawa, and R. Morris. 2008. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982-2003). *Marine Biology* 154(5):887-898.
- Chaloupka, M. Y., and C. J. Limpus. 1997. Robust statistical modelling of hawksbill sea turtle growth rates (southern Great Barrier Reef). *Marine Ecology Progress Series* 146(1-3):1-8.
- Chaloupka, M. Y., and J. A. Musick. 1997. Age growth and population dynamics. Pages 233-276 in P. L. Lutz, and J. A. Musick, editors. *The Biology of Sea Turtles*. CRC Press, Boca Raton, Florida.
- Chiappone, M., H. Dienes, D. W. Swanson, and S. L. Miller. 2005. Impacts of lost fishing gear on coral reef sessile invertebrates in the Florida Keys National Marine Sanctuary. *Biological Conservation* 121(2):221-230.
- Chytalo, K. 1996. Summary of Long Island Sound Dredging Windows Strategy Workshop. Management of Atlantic Coastal Marine Fish Habitat: Proceedings of a workshop for habitat managers. ASMFC Habitat Management Series #2. Atlantic States Marine Fisheries Commission.
- CITES. 2010. Consideration of proposals for amendment of appendices I and II (CoP15 Prop. 15). 15th meeting of the Conference of the Parties, Doha (Qatar). CITES.
- Coelho, R., and coauthors. 2009. Notes of the reproduction of the oceanic whitetip shark, *Carcharhinus longimanus*, in the southwestern equatorial Atlantic Ocean. . Pages 1734-1740 in.
- Colin, P. L. 1992. Reproduction of the Nassau grouper, *Epinephelus striatus* (Pisces: Serranidae) and its relationship to environmental conditions. *Environmental Biology of Fishes* (34):357-377.
- Colin, P. L., W. A. Laroche, and E. B. Brothers. 1997. Ingress and settlement in the Nassau grouper, *Epinephelus striatus* (Pisces: Serranidae), with relationship to spawning occurrence. *Bulletin of Marine Science* 60(3):656-667.
- Collazo, J. A., R. Boulan, and T. L. Tallevast. 1992. Abundance and growth patterns of *Chelonia mydas* in Culebra, Puerto Rico. *Journal of Herpetology* 26(3):293-300.
- Collette, B., and G. Klein-MacPhee. 2002. *Fishes of the Gulf of Maine*, 3rd edition. Smithsonian Institution Press.
- Collins, M. R., S. G. Rogers, and T. I. J. Smith. 1996. Bycatch of sturgeons along the southern Atlantic coast of the USA. *North American Journal of Fisheries Management* 16(1):24-29.

- Collins, M. R., S. G. Rogers, T. I. J. Smith, and M. L. Moser. 2000a. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. *Bulletin of Marine Science* 66(3):917-928.
- Collins, M. R., and T. I. J. Smith. 1997. Distributions of shortnose and Atlantic sturgeons in South Carolina. *North American Journal of Fisheries Management* 17(4):955-1000.
- Collins, M. R., T. I. J. Smith, W. C. Post, and O. Pashuk. 2000b. Habitat Utilization and Biological Characteristics of Adult Atlantic Sturgeon in Two South Carolina Rivers. *Transactions of the American Fisheries Society* 129(4):982-988.
- Compagno, L. J. V. 1984. Part 2. Carcharhiniformes. Pages 251-655 in *FAO Species Catalogue. Sharks of the World. An Annotated and Illustrated Catalogue of Sharks Species Known to Date*, volume 4. FAO.
- Conant, T. A., and coauthors. 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.
- Cooper, K. 1989. Effects of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans on aquatic organisms. *Reviews in Aquatic Sciences* 1(2):227-242.
- Corsolini, S., S. Aurigi, and S. Focardi. 2000. Presence of polychlorobiphenyls (PCBs) and coplanar congeners in the tissues of the Mediterranean loggerhead turtle *Caretta caretta*. *Marine Pollution Bulletin* 40(11):952-960.
- Cortés, E. 1999. Standardized diet compositions and trophic levels of sharks. *ICES Journal of Marine Science: Journal du Conseil* 56(5):707-717.
- Cortés, E., and coauthors. 2010. Ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. . *Aquatic Living Resources* 23:25-34.
- Cortés, E., C. A. Brown, and L. R. Beerkircher. 2007. Relative Abundance of Pelagic Sharks in the Western North Atlantic Ocean, Including the Gulf of Mexico and Caribbean Sea. . *Gulf and Caribbean Research* 19(2):37-52.
- Cortés, E., and coauthors. 2012. Expanded ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. .
- Crabbe, M. J. 2008. Climate change, global warming and coral reefs: modelling the effects of temperature. *Computational Biology and Chemistry* 32(5):311-4.
- Crance, J. H. 1987. Habitat suitability index curves for anadromous fishes. In: *Common strategies of anadromous and catadromous fishes: proceedings of an International Symposium held in Boston, Massachusetts, USA, March 9-13, 1986*. Pages 554 in M. J. Dadswell, editor. American Fisheries Society, Bethesda, Maryland.

- Crouse, D. T. 1999. Population modeling and implications for Caribbean hawksbill sea turtle management *Chelonian Conservation and Biology* 3(2):185-188.
- Crouse, D. T., L. B. Crowder, and H. Caswell. 1987. A Stage-Based Population Model for Loggerhead Sea Turtles and Implications for Conservation. *Ecology* 68(5):1412-1423.
- Crowder, L. B., D. T. Crouse, S. S. Heppell, and T. H. Martin. 1994. Predicting the impact of turtle excluder devices on loggerhead sea turtle populations. *Ecological Applications* 4(3):437-445.
- D'Ilio, S., D. Mattei, M. F. Blasi, A. Alimonti, and S. Bogialli. 2011. The occurrence of chemical elements and POPs in loggerhead turtles (*Caretta caretta*): An overview. *Marine Pollution Bulletin* 62(8):1606-1615.
- Dadswell, M. J. 2006. A review of the status of Atlantic sturgeon in Canada, with comparisons to populations in the United States and Europe. *Fisheries* 31(5):218-229.
- Dahl, T. E., and C. E. Johnson. 1991. Status and trends of wetlands in the conterminous United States, mid-1970s to mid-1980s. U.S. Fish and Wildlife Service, Washington, D.C.
- Damon-Randall, K., and coauthors. 2010. Atlantic sturgeon research techniques. NOAA technical memorandum NMFS-NE ; 215. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Region, Northeast Fisheries Science Center, Woods Hole, Mass.
- Daniels, R. C., T. W. White, and K. K. Chapman. 1993. Sea-level rise - destruction of threatened and endangered species habitat in South Carolina. *Environmental Management* 17(3):373-385.
- Davenport, J., D. L. Holland, and J. East. 1990. Thermal and biochemical characteristics of the lipids of the leatherback turtle (*Dermochelys coriacea*): Evidence of endothermy. *Journal of the Marine Biological Association of the United Kingdom* 70:33-41.
- de Silva, J. A., R. E. Condrey, and B. A. Thompson. 2001. Profile of shark bycatch in the US Gulf of Mexico menhaden fishery. *North American Journal of Fisheries Management* 21(1):111-124.
- Dickerson, D. 2011. Observed takes of sturgeon and turtles from dredging operations along the Atlantic Coast. U.S. Army Engineer Research and Development Center Environmental Laboratory, Vicksburg, MS.
- Diemer, K. M., B. Q. Mann, and N. E. Hussey. 2011. Distribution and movement of scalloped hammerhead *Sphyrna lewini* and smooth hammerhead *Sphyrna zygaena*

- sharks along the east coast of southern Africa. *African Journal of Marine Science* 33(2):229-238.
- Diez, C. E., and R. P. Van Dam. 2002. Habitat effect on hawksbill turtle growth rates on feeding grounds at Mona and Monito Islands, Puerto Rico. *Marine Ecology Progress Series* 234:301-309.
- Diez, C. E., and R. P. Van Dam. 2007. In-water surveys for marine turtles at foraging grounds of Culebra Archipelago, Puerto Rico
- Dodd Jr., C. K. 1988. Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). U.S. Fish and Wildlife Service, 88(14).
- Doughty, R. W. 1984. Sea turtles in Texas: A forgotten commerce. *Southwestern Historical Quarterly* 88:43-70.
- Dovel, W. L., and T. J. Berggren. 1983. Atlantic sturgeon of the Hudson Estuary, New York. *New York Fish and Game Journal* 30(2):140-172.
- Dow, W., K. Eckert, M. Palmer, and P. Kramer. 2007. An atlas of sea turtle nesting habitat for the wider Caribbean region. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy, Beaufort, North Carolina.
- Drevnick, P. E., and M. B. Sandheinrich. 2003. Effects of dietary methylmercury on reproductive endocrinology of fathead minnows. *Environmental Science and Technology* 37(19):4390-4396.
- Dudley, S. F. J., and C. A. Simpfendorfer. 2006. Population status of 14 shark species caught in the protective gillnets off KwaZulu–Natal beaches, South Africa, 1978–2003. *Marine and Freshwater Research* 57(2):225-240.
- Duncan, K. M., and K. N. Holland. 2006. Habitat use, growth rates and dispersal patterns of juvenile scalloped hammerhead sharks *Sphyrna lewini* in a nursery habitat. *Marine Ecology Progress Series* 312:211-221.
- Duncan, W. W., M. C. Freeman, C. A. Jennings, and J. T. McLean. 2003. Considerations for flow alternatives that sustain Savannah River fish populations. Pages 4 *in* K. J. Hatcher, editor Georgia Water Resources Conference. Institute of Ecology, The University of Georgia, Athens, Georgia.
- Dunton, K. J., A. Jordaan, K. A. McKown, D. O. Conover, and M. G. Frisk. 2010. Abundance and distribution of Atlantic sturgeon (*Acipenser oxyrinchus*) within the Northwest Atlantic Ocean, determined from five fishery-independent surveys. *Fishery Bulletin* 108(4):450-465.
- Duque, V. M., V. M. Paez, and J. A. Patino. 2000. Ecología de anidación y conservación de la tortuga cana, *Dermochelys coriacea*, en la Playona, Golfo de Uraba Chocoano (Colombia), en 1998 *Actualidades Biologicas Medellín* 22(72):37-53.

- Dutton, D. L., P. H. Dutton, M. Chaloupka, and R. H. Boulon. 2005. Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. *Biological Conservation* 126(2):186-194.
- DWH Trustees. 2015. DWH Trustees (Deepwater Horizon Natural Resource Damage Assessment Trustees). 2015. Deepwater Horizon Oil Spill: Draft Programmatic Damage Assessment and Restoration Plan and Draft Programmatic Environmental Impact Statement. Retrieved from <http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan/>.
- Dwyer, F. J., D. K. Hardesty, C. G. Ingersoll, J. L. Kunz, and D. W. Whites. 2000. Assessing contaminant sensitivity of American shad, Atlantic sturgeon and shortnose sturgeon. Final report - February 2000. U.S. Geological Survey, Columbia Environmental Research Center Columbia, Missouri.
- Dwyer, K. L., C. E. Ryder, and R. Prescott. 2003. Anthropogenic mortality of leatherback turtles in Massachusetts waters. Pages 260 in J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation, Miami, Florida.
- Eckert, K. L. 1995. Hawksbill sea turtle (*Eretmochelys imbricata*). Pages 76-108 in National Marine Fisheries Service and U.S. Fish and Wildlife Service Status Reviews for Sea Turtles Listed under the Endangered Species Act of 1973. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Springs, Maryland.
- Eckert, K. L., and S. A. Eckert. 1990. Embryo mortality and hatch success in (*in situ*) and translocated leatherback sea turtle (*Dermochelys coriacea*) eggs. *Biological Conservation* 53:37-46.
- Eckert, K. L., S. A. Eckert, T. W. Adams, and A. D. Tucker. 1989. Inter-nesting migrations by leatherback sea turtles (*Dermochelys coriacea*) in the West Indies. *Herpetologica* 45(2):190-194.
- Eckert, K. L., J. A. Overing, and B. B. Lettsome. 1992. Sea turtle recovery action plan for the British Virgin Islands. UNEP Caribbean Environment Programme, Wider Caribbean Sea Turtle Recovery Team and Conservation Network, Kingston, Jamaica.
- Eckert, K. L., B. P. Wallace, J. G. Frazier, S. A. Eckert, and P. C. H. Pritchard. 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Fish and Wildlife Service.
- Eckert, S. A. 1989. Diving and foraging behavior of the leatherback sea turtle, *Dermochelys coriacea*. University of Georgia, Athens, Georgia.
- Eckert, S. A. 2006. High-use oceanic areas for Atlantic leatherback sea turtles (*Dermochelys coriacea*) as identified using satellite telemetered location and dive information. *Marine Biology* 149(5):1257-1267.

- Eckert, S. A., and coauthors. 2006. Internesting and postnesting movements and foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. *Chelonian Conservation and Biology* 5(2):239-248.
- Eckert, S. A., D. W. Nellis, K. L. Eckert, and G. L. Kooyman. 1984. Deep diving record for leatherbacks. *Marine Turtle Newsletter* 31:4.
- Eckert, S. A., and L. Sarti. 1997. Distant fisheries implicated in the loss of the world's largest leatherback nesting population. *Marine Turtle Newsletter* 78:2-7.
- Edwards, R. E., F. M. Parauka, and K. J. Sulak. 2007. New insights into marine migration and winter habitat of Gulf sturgeon. *American Fisheries Society Symposium* 57:14.
- Eggleston, D. B. 1995. Recruitment in Nassau grouper *Epinephelus striatus*: post-settlement abundance, microhabitat features and ontogenetic habitat shifts. *Marine Ecology Progress Series* 124:9-22.
- Eguchi, T., P. H. Dutton, S. A. Garner, and J. Alexander-Garner. 2006. Estimating juvenile survival rates and age at first nesting of leatherback turtles at St. Croix, U.S. Virgin Islands. Pages 292-293 in M. Frick, A. Panagopoulou, A. F. Rees, and K. Williams, editors. *Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation*. International Sea Turtle Society, Athens, Greece.
- Ehrhart, L. M. 1983. Marine turtles of the Indian River Lagoon System. *Florida Scientist* 46(3/4):337-346.
- Ehrhart, L. M., W. E. Redfoot, and D. A. Bagley. 2007. Marine turtles of the central region of the Indian River Lagoon System, Florida. *Florida Scientist* 70(4):415-434.
- Ehrhart, L. M., and R. G. Yoder. 1978. Marine turtles of Merritt Island National Wildlife Refuge, Kennedy Space Centre, Florida. *Florida Marine Research Publications* 33:25-30.
- Enzenauer, M. P., B. M. Deacy, and J. K. Carlson. 2016. Characterization of the Shark Bottom Longline Fishery, 2015. NOAA Technical Memorandum NMFS-SEFSC-689.
- EPA. 2005. National Coastal Condition Report II. U.S. Environmental Protection Agency.
- EPA. 2012. Climate Change. www.epa.gov/climatechange/index.html.
- Epperly, S. P., and C. Boggs. 2004. Post-hooking mortality in pelagic longline fisheries using "J" hooks and circle hooks. Application of new draft criteria to data from the Northeast Distant Experiments in the Atlantic. Contribution # PRD-03/04-04 of NOAA, National Marine Fisheries Service, Miami, FL, USA.
- Epperly, S. P., J. Braun-McNeill, and P. M. Richards. 2007. Trends in catch rates of sea turtles in North Carolina, USA. *Endangered Species Research* 3(3):283-293.

- Epperly, S. P., and W. G. Teas. 2002. Turtle excluder devices - Are the escape openings large enough? *Fishery Bulletin* 100(3):466-474.
- Erickson, D. L., and coauthors. 2011. Use of pop-up satellite archival tags to identify oceanic-migratory patterns for adult Atlantic Sturgeon, *Acipenser oxyrinchus oxyrinchus* Mitchell, 1815. *Journal of Applied Ichthyology* 27(2):356-365.
- Evermann, B. W., and B. A. Bean. 1897. Report on the Fisheries of Indian River, Florida. United States Commission of Fish and Fisheries, Washington D.C.
- FAO. 2010. Report of the third FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species. Rome, 7–12 December 2009, Rome.
- Farrae, D. J., W. C. Post, and T. L. Darden. 2017. Genetic characterization of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, in the Edisto River, South Carolina and identification of genetically discrete fall and spring spawning. *Conservation Genetics*:1-11.
- Fernandes, S. J., G. B. Zydlewski, J. D. Zydlewski, G. S. Wippelhauser, and M. T. Kinnison. 2010. Seasonal distribution and movements of shortnose sturgeon and Atlantic sturgeon in the Penobscot River Estuary, Maine. *Transactions of the American Fisheries Society* 139(5):1436-1449.
- Fernandez-Carvalho, J., R. Coelho, M. N. Santos, and S. Amorim. 2015. Effects of hook and bait in a tropical northeast Atlantic pelagic longline fishery: Part II—Target, bycatch and discard fishes. *Fisheries Research* 164:312-321.
- Ferraroli, S., J. Y. Georges, P. Gaspar, and Y. Le Maho. 2004. Where leatherback turtles meet fisheries. *Nature* 429:521-522.
- Fish, M. R., and coauthors. 2005. Predicting the Impact of Sea-Level Rise on Caribbean Sea Turtle Nesting Habitat. *Conservation Biology* 19(2):482-491.
- Fisher, M. 2009. Atlantic Sturgeon Progress Report. Delaware State Wildlife Grant, Project T 4-1. December 16, 2008 to December 15, 2009.
- Fisher, M. 2011. Atlantic Sturgeon Progress Report. Delaware State Wildlife Grant, Project T 4-1, October 1, 2006 to October 15, 2010.
- FitzSimmons, N. N., L. W. Farrington, M. J. McCann, C. J. Limpus, and C. Moritz. 2006. Green turtle populations in the Indo-Pacific: A (genetic) view from microsatellites. Pages 111 *in* N. Pilcher, editor Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation.
- Fleming, E. H. 2001. *Swimming Against the Tide: Recent Surveys of Exploitation, Trade, And Management of Marine Turtles In the Northern Caribbean*. TRAFFIC North America, Washington, D.C., USA.

- Foley, A. M., and coauthors. 2017. Distributions, relative abundances, and mortality factors of sea turtles in Florida during 1980–2014 as determined from strandings. Part of final grant report submitted to the Department of Commerce/National Oceanographic and Atmospheric Administration as a product of award # NA13NMF4720046 (Florida Fish and Wildlife Conservation Commission File Code: F4115-13-F). 158 pp.
- Foley, A. M., B. A. Schroeder, and S. L. MacPherson. 2008a. Post-nesting migrations and resident areas of Florida loggerheads (*Caretta caretta*). Pages 75-76 in H. J. Kalb, A. S. Rhode, K. Gayheart, and K. Shanker, editors. Twenty-Fifth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Savannah, Georgia.
- Foley, A. M., B. A. Schroeder, A. E. Redlow, K. J. Fick-Child, and W. G. Teas. 2005. Fibropapillomatosis in stranded green turtles (*Chelonia mydas*) from the eastern United States (1980-98): Trends and associations with environmental factors. *Journal of Wildlife Diseases* 41(1):29-41.
- Foley, A. M., and coauthors. 2008b. Distributions, Relative Abundances, and Mortality Factors for Sea Turtles in Florida from 1980 through 2007 as Determined From Strandings. A final Report Submitted to NMFS. NMFS.
- Foley, A. M., and coauthors. 2007. Characteristics of a green turtle (*Chelonia mydas*) assemblage in northwestern Florida determined during a hypothermic stunning event. *Gulf of Mexico Science* 25(2):131-143.
- Foley, K. 2007. First Confirmed Redcord of Nassau Grouper *Epinephelus striatus* (Pisces: Serranidae) in the Flower Garden Banks National Marine Sanctuary. *Gulf of Mexico Science*, 2007 (32), pp. 162-165. .
- Formia, A. 1999. Les tortues marines de la Baie de Corisco. *Canopee* 14: i-ii.
- Francis, M. P., L. H. Griggs, and S. J. Baird. 2001. Pelagic shark bycatch in the New Zealand tuna longline fishery. *Marine and Freshwater Research* 52(2):165-178.
- Frankham, R., C. J. A. Bradshaw, and B. W. Brook. 2014. Genetics in conservation management: revised recommendations for the 50/500 rules, Red List criteria and population viability analyses. *Biological Conservation* 170:56-63.
- Frazer, N. B., and L. M. Ehrhart. 1985. Preliminary growth models for green, (*Chelonia mydas*) and loggerhead, (*Caretta caretta*), turtles in the wild. *Copeia* 1985(1):73-79.
- Frédou, F. L., and coauthors. 2015. Sharks caught by the Brazilian tuna longline fleet: an overview. *Rev. Fish Biol. Fish.* 25:365-377.
- Fretey, J. 2001. Biogeography and conservation of marine turtles of the Atlantic Coast of Africa, UNEbraskaP/CMississippi Secretariat.

- Fretey, J., A. Billes, and M. Tiwari. 2007. Leatherback, *Dermochelys coriacea*, nesting along the Atlantic coast of Africa. *Chelonian Conservation and Biology* 6(1):126-129.
- Gallagher, A. J., J. E. Serafy, S. J. Cooke, and N. Hammerschlag. 2014. Physiological stress response, reflex impairment, and survival of five sympatric shark species following experimental capture and release. *Marine Ecology Progress Series* 496:207-218.
- Garcia-Parraga, D., and coauthors. 2014. Decompression sickness ('the bends') in sea turtles. *Dis Aquat Organ* 111(3):191-205.
- Garcia M., D., and L. Sarti. 2000. Reproductive cycles of leatherback turtles. Pages 163 in F. A. Abreu-Grobois, R. Briseno-Duenas, R. Marquez, and L. Sarti, editors. Eighteenth International Sea Turtle Symposium.
- Garduño-Andrade, M., V. Guzmán, E. Miranda, R. Briseño-Dueñas, and F. A. Abreu-Grobois. 1999. Increases in hawksbill turtle (*Eretmochelys imbricata*) nestings in the Yucatán Peninsula, Mexico, 1977-1996: Data in support of successful conservation? *Chelonian Conservation and Biology* 3(2):286-295.
- Garrett, C. 2004. Priority Substances of Interest in the Georgia Basin - Profiles and background information on current toxics issues. Canadian Toxics Work Group Puget Sound, Georgia Basin International Task Force, GBAP Publication No. EC/GB/04/79.
- Gearhart, J. L. 2010. Evaluation of a turtle excluder device (TED) designed for use in the U.S. mid-Atlantic croaker fishery. NOAA Technical Memorandum NMFS-SEFSC-606.
- Geraci, J. R. 1990. Physiologic and toxic effects on cetaceans. Pages 167-197 in J. R. Geraci, and D. J. S. Aubin, editors. *Sea Mammals and Oil: Confronting the Risks*. Academic Press, San Diego.
- Giesy, J. P., J. Newsted, and D. L. Garling. 1986. Relationships Between Chlorinated Hydrocarbon Concentrations and Rearing Mortality of Chinook Salmon (*Oncorhynchus Tshawytscha*) Eggs from Lake Michigan. *Journal of Great Lakes Research* 12(1):82-98.
- Gilbert, C. R. 1989. Species profiles : life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic Bight) : Atlantic and shortnose sturgeons. Coastal Ecology Group, Waterways Experiment Station, U.S. Dept. of the Interior, Fish and Wildlife Service, Research and Development, National Wetlands Research Center, Vicksburg, MS, Washington, DC.
- Gilman, E., and coauthors. 2006. Reducing sea turtle by-catch in pelagic longline fisheries. *Fish and Fisheries* 7(1):2-23.

- Gilman, E. L., J. Ellison, N. C. Duke, and C. Field. 2008. Threats to mangroves from climate change and adaptation options: A review. *Aquatic Botany* 89(2):237-250.
- Gilmore, G. R. 1995. Environmental and Biogeographic Factors Influencing Ichthyofaunal Diversity: Indian River Lagoon. *Bulletin of Marine Science* 57(1):153-170.
- Girard, C., A. D. Tucker, and B. Calmettes. 2009. Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: Dispersal in highly dynamic conditions. *Marine Biology* 156(9):1827-1839.
- Gladys Porter Zoo. 2013. Gladys Porter Zoo's Preliminary Annual Report on the Mexico/United States of America Population Restoration Project for the Kemp's Ridley Sea Turtle, *Lepidochelys kempii*, on the Coasts of Tamaulipas, Mexico 2013.
- Glen, F., A. C. Broderick, B. J. Godley, and G. C. Hays. 2006. Thermal control of hatchling emergence patterns in marine turtles. *Journal of Experimental Marine Biology and Ecology* 334(1):31-42.
- Goff, G. P., and J. Lien. 1988. Atlantic leatherback turtles, *Dermochelys coriacea*, in cold water off Newfoundland and Labrador. *Canadian Field-Naturalist* 102:1-5.
- Gonzalez Carman, V., and coauthors. 2011. Argentinian coastal waters: A temperate habitat for three species of threatened sea turtles. *Marine Biology Research* 7:500-508.
- Goodyear, C. P. 1993. Spawning stock biomass per recruit in fisheries management: foundation and current use. *Canadian Special Publication of Fisheries and Aquatic Sciences*:67-82.
- Graham, T. R. 2009. Scyphozoan jellies as prey for leatherback sea turtles off central California. Master's Theses. San Jose State University.
- Grant, S. C. H., and P. S. Ross. 2002. Southern Resident killer whales at risk: Toxic chemicals in the British Columbia and Washington environment. Department of Fisheries and Oceans Canada, Sidney, B.C.
- Green, D. 1993. Growth rates of wild immature green turtles in the Galápagos Islands, Ecuador. *Journal of Herpetology* 27(3):338-341.
- Greene, K. E., J. L. Zimmerman, R. W. Laney, and J. C. Thomas-Blate. 2009. Atlantic coast diadromous fish habitat: A review of utilization, threats, recommendations for conservation, and research needs. Atlantic States Marine Fisheries Commission Washington, D.C.
- Greer, A. E. J., J. D. J. Lazell, and R. M. Wright. 1973. Anatomical evidence for a counter-current heat exchanger in the leatherback turtle (*Dermochelys coriacea*). *Nature* 244:181.

- Groombridge, B. 1982. Kemp's ridley or Atlantic ridley, *Lepidochelys kempii* (Garman 1980). The IUCN Amphibia, Reptilia Red Data Book:201-208.
- Groombridge, B., and R. Luxmoore. 1989. The Green Turtle and Hawksbill (Reptilia: Cheloniidae): World Status, Exploitation and Trade. Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, Lausanne, Switzerland.
- Grunwald, C., L. Maceda, J. Waldman, J. Stabile, and I. Wirgin. 2008. Conservation of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*: Delineation of stock structure and distinct population segments. *Conservation Genetics* 9(5):1111-1124.
- Guilbard, F., J. Munro, P. Dumont, D. Hatin, and R. Fortin. 2007. Feeding ecology of Atlantic sturgeon and lake sturgeon co-occurring in the St. Lawrence estuarine transition zone. *American Fisheries Society Symposium* 56:85.
- Guseman, J. L., and L. M. Ehrhart. 1992. Ecological geography of western Atlantic loggerheads and green turtles: Evidence from remote tag recoveries. Pages 50 in M. Salmon, and J. Wyneken, editors. Eleventh Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Jekyll Island, Georgia.
- GWC. 2006. Georgia Water Coalition. Interbasin Transfer Fact Sheet. <http://www.garivers.org/gawater/pdf%20files/IBT%20fact%20sheet02-06.pdf>.
- Hager, C., J. Kahn, C. Watterson, J. Russo, and K. Hartman. 2014. Evidence of Atlantic sturgeon spawning in the York River system. *Transactions of the American Fisheries Society* 143(5):1217-1219.
- Hale, E. A., and coauthors. 2016. Abundance Estimate for and Habitat Use by Early Juvenile Atlantic Sturgeon within the Delaware River Estuary. *Transactions of the American Fisheries Society* 145(6):1193-1201.
- Hale, L. F., and J. Carlson. 2007. Characterization of the shark bottom longline fishery, 2005-2006. NOAA Technical Memorandum NMFS-SEFSC-554, 28 p. .
- Hale, L. F., S. J. B. Gulak, and J. K. Carlson. 2010. Characterization of the Shark Bottom Longline Fishery: 2009. U.S. Department of Commerce.
- Hammerschmidt, C. R., M. B. Sandheinrich, J. G. Wiener, and R. G. Rada. 2002. Effects of dietary methylmercury on reproduction of fathead minnows. *Environmental Science and Technology* 36(5):877-883.
- Hare, J. A., and coauthors. 2016. A Vulnerability Assessment of Fish and Invertebrates to Climate Change on the Northeast US Continental Shelf. *PLoS ONE* 11(2):e0146756.

- Harry, A. V., W. G. Macbeth, A. N. Gutteridge, and C. A. Simpfendorfer. 2011a. The life histories of endangered hammerhead sharks (Carcharhiniformes, Sphyrnidae) from the east coast of Australia. *Journal of Fish Biology* 78(7):2026-2051.
- Harry, A. V., and coauthors. 2011b. Evaluating catch and mitigating risk in a multispecies, tropical, inshore shark fishery within the Great Barrier Reef World Heritage Area. *Marine and Freshwater Research* 62(6):710-721.
- Hart, K. M., M. M. Lamont, I. Fujisaki, A. D. Tucker, and R. R. Carthy. 2012. Common coastal foraging areas for loggerheads in the Gulf of Mexico: Opportunities for marine conservation. *Biological Conservation* 145:185-194.
- Hartwell, S. I. 2004. Distribution of DDT in sediments off the central California coast. *Marine Pollution Bulletin* 49(4):299-305.
- Hatin, D., J. Munro, F. Caron, and R. Simons. 2007. Movements, home range size, and habitat use and selection of early juvenile Atlantic sturgeon in the St. Lawrence estuarine transition zone. Pages 129 *in* American Fisheries Society Symposium. American Fisheries Society.
- Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. *Global Change Biology* 13:1-10.
- Hayes, C. G., Y. Jiao, and E. Cortés. 2009. Stock assessment of scalloped hammerheads in the western North Atlantic Ocean and Gulf of Mexico. *North American Journal of Fisheries Management* 29(5):1406-1417.
- Hays, G. C., and coauthors. 2001. The diving behavior of green turtles undertaking oceanic migration to and from Ascension Island: Dive durations, dive profiles, and depth distribution. *Journal of Experimental Biology* 204:4093-4098.
- Hays, G. C., and coauthors. 2002. Water temperature and internesting intervals for loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtles. *Journal of Thermal Biology* 27(5):429-432.
- Hays, G. C., J. D. R. Houghton, and A. E. Myers. 2004. Pan-Atlantic leatherback turtle movements. *Nature* 429:522.
- Hazel, J., I. R. Lawler, H. Marsh, and S. Robinson. 2007. Vessel speed increases collision risk for the green turtle *Chelodina mydas*. *Endangered Species Research* 3:105-113.
- Hazin, F., A. Fischer, and M. Broadhurst. 2001. Aspects of reproductive biology of the scalloped hammerhead shark, *Sphyrna lewini*, off northeastern Brazil. *Environmental Biology of Fishes* 61(2):151-159.

- Hearn, A., J. T. Ketchum, A. P. Klimley, E. Espinoza, and C. Penaherrera. 2010. Hotspots within hotspots? Hammerhead shark movements around Wolf Island, Galapagos Marine Reserve. *Marine Biology* 157:1899-1915.
- Heemstra, P. C., and J. E. Randall. 1993. Groupers of the World (Family Serranidae, Subfamily Epinephelinae): An Annotated and Illustrated Catalogue of the Grouper, Rockcod, Hind, Coral Grouper and Lyretail Species Known to Date. FAO Species Catalog. Vol. 16, No. 125. Food and Agriculture Organization of the United Nations, Rome.
- Heppell, S. S., and coauthors. 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. *Chelonian Conservation and Biology* 4(4):767-773.
- Heppell, S. S., L. B. Crowder, D. T. Crouse, S. P. Epperly, and N. B. Frazer. 2003a. Population models for Atlantic loggerheads: Past, present, and future. Pages 255-273 in A. Bolten, and B. Witherington, editors. *Loggerhead Sea Turtles*. Smithsonian Books, Washington, D. C.
- Heppell, S. S., L. B. Crowder, and T. R. Menzel. 1999. Life table analysis of long-lived marine species with implications for conservation and management. Pages 137-148 in *American Fisheries Society Symposium*.
- Heppell, S. S., M. L. Snover, and L. Crowder. 2003b. Sea turtle population ecology. Pages 275-306 in P. Lutz, J. A. Musick, and J. Wyneken, editors. *The Biology of Sea Turtles*. CRC Press, Boca Raton, Florida.
- Herbst, L. H. 1994. Fibropapillomatosis of marine turtles. *Annual Review of Fish Diseases* 4:389-425.
- Herbst, L. H., and coauthors. 1995. An infectious etiology for green turtle fibropapillomatosis. *Proceedings of the American Association for Cancer Research Annual Meeting* 36:117.
- Hildebrand, H. H. 1963. Hallazgo del area de anidacion de la tortuga marina "lora", *Lepidochelys kempi* (Garman), en la costa occidental del Golfo de Mexico (Rept., Chel.). *Ciencia, Mexico* 22:105-112.
- Hildebrand, H. H. 1982. A historical review of the status of sea turtle populations in the western Gulf of Mexico. Pages 447-453 in K. A. Bjorndal, editor. *Biology and Conservation of Sea Turtles*. Smithsonian Institution Press, Washington, D. C.
- Hill, R. L., and Y. Sadovy de Mitcheson. 2013. Nassau Grouper, *Epinephelus striatus* (Bloch 1792), Status Review Document. Report to National Marine Fisheries Service, Southeast Regional Office. .

- Hillis, Z.-M., and A. L. Mackay. 1989. Research report on nesting and tagging of hawksbill sea turtles *Eretmochelys imbricata* at Buck Island Reef National Monument, U.S. Virgin Islands, 1987-88.
- Hilterman, M., E. Goverse, M. Godfrey, M. Girondot, and C. Sakimin. 2003. Seasonal sand temperature profiles of four major leatherback nesting beaches in the Guyana Shield. Pages 189-190 in J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation.
- Hirth, H., J. Kasu, and T. Mala. 1993. Observations on a leatherback turtle *Dermochelys coriacea* nesting population near Piguwa, Papua New Guinea. Biological Conservation 65:77-82.
- Hirth, H. F. 1971. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus) 1758. Food and Agriculture Organization.
- Hirth, H. F. 1997. Synopsis of the biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). Biological Report 91(1):120.
- Hirth, H. F., and E. M. A. Latif. 1980. A nesting colony of the hawksbill turtle (*Eretmochelys imbricata*) on Seil Ada Kebir Island, Suakin Archipelago, Sudan. Biological Conservation 17:125-130.
- Holland, K. N., B. M. Wetherbee, J. D. Peterson, and C. G. Lowe. 1993. Movements and distribution of hammerhead shark pups on their natal grounds. Copeia (2):495-502.
- Holton, J. W. J., and J. B. Walsh. 1995. Long-term dredged material management plan for the upper James River, Virginia. Waterway Surveys and Engineering, Ltd, Virginia Beach, VA.
- Houghton, J. D. R., T. K. Doyle, M. W. Wilson, J. Davenport, and G. C. Hays. 2006. Jellyfish aggregations and leatherback turtle foraging patterns in a temperate coastal environment. Ecology 87(8):1967-1972.
- Howey-Jordan, L. A., and coauthors. 2013. Complex Movements, Philopatry and Expanded Depth Range of a Severely Threatened Pelagic Shark, the Oceanic Whitetip (*Carcharhinus longimanus*) in the Western North Atlantic. PLoS ONE 8(2):e56588.
- Howey, L. A., and coauthors. 2016. Into the deep: the functionality of mesopelagic excursions by an oceanic apex predator. Ecology and Evolution 6(15):5290-5304.
- Hughes, G. R. 1996. Nesting of the leatherback turtle (*Dermochelys coriacea*) in Tongaland, KwaZulu-Natal, South Africa, 1963-1995. Chelonian Conservation Biology 2(2):153-158.

- Ingram, E. C., and D. L. Peterson. 2016. Annual Spawning Migrations of Adult Atlantic Sturgeon in the Altamaha River, Georgia. *Marine and Coastal Fisheries* 8(1):595-606.
- IOTC. 2014. Report of the Seventeenth Session of the IOTC Scientific Committee. .
- IPCC. 2007. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.
- IPCC. 2008. Climate Change and Water, Technical Paper of the Intergovernmental Panel on Climate Change, IPCC Secretariat, Geneva, 210 pp.
- ISED. 2014. International Sawfish Encounter Database. Florida Museum of Natural History, Gainesville, Florida.
<http://www.flmnh.ufl.edu/fish/sharks/sawfish/sawfishdatabase.html>.
- Iwata, H., S. Tanabe, N. Sakai, and R. Tatsukawa. 1993. Distribution of persistent organochlorines in the oceanic air and surface seawater and the role of ocean on their global transport and fate. *Environmental Science and Technology* 27(6):1080-1098.
- Jacobson, E. R. 1990. An update on green turtle fibropapilloma. *Marine Turtle Newsletter* 49:7-8.
- Jacobson, E. R., and coauthors. 1989. Cutaneous fibropapillomas of green turtles (*Chelonia mydas*). *Journal Comparative Pathology* 101:39-52.
- Jacobson, E. R., S. B. Simpson Jr., and J. P. Sundberg. 1991. Fibropapillomas in green turtles. Pages 99-100 in G. H. Balazs, and S. G. Pooley, editors. *Research Plan for Marine Turtle Fibropapilloma*, volume NOAA-TM-NMFS-SWFSC-156.
- James, M. C., S. A. Eckert, and R. A. Myers. 2005. Migratory and reproductive movements of male leatherback turtles (*Dermochelys coriacea*). *Marine Biology* 147(4):845-853.
- James, M. C., S. A. Sherrill-Mix, and R. A. Myers. 2007. Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. *Marine Ecology Progress Series* 337:245-254.
- Jiao, Y., E. Cortés, K. Andrews, and F. Guo. 2011. Poor-data and data-poor species stock assessment using a Bayesian hierarchical approach. *Ecological Applications* 21(7):2691-2708.
- Johnson, S. A., and L. M. Ehrhart. 1994. Nest-site fidelity of the Florida green turtle. Pages 83 in B. A. Schroeder, and B. E. Witherington, editors. *Thirteenth Annual Symposium on Sea Turtle Biology and Conservation*.

- Johnson, S. A., and L. M. Ehrhart. 1996. Reproductive ecology of the Florida green turtle: Clutch frequency. *Journal of Herpetology* 30(3):407-410.
- Jones, T. T., M. D. Hastings, B. L. Bostrom, D. Pauly, and D. R. Jones. 2011. Growth of captive leatherback turtles, *Dermochelys coriacea*, with inferences on growth in the wild: Implications for population decline and recovery. *Journal of Experimental Marine Biology and Ecology* 399(1):84-92.
- Jorgensen, E. H., O. Aas-Hansen, A. G. Maule, J. E. T. Strand, and M. M. Vijayan. 2004a. PCB impairs smoltification and seawater performance in anadromous Arctic charr (*Salvelinus alpinus*). *Comparative Biochemistry and Physiology C Toxicology and Pharmacology* 138(2):203-212.
- Jorgensen, E. H., O. Aas-Hansen, A. G. Maule, J. E. T. Strand, and M. M. Vijayan. 2004b. PCB impairs smoltification and seawater performance in anadromous Arctic charr (*Salvelinus alpinus*). *Comparative Biochemistry and Physiology Part C Toxicology & Pharmacology* 138(2):203-212.
- Jorgensen, S. J., A. P. Klimley, and A. F. Muhlia-Melo. 2009. Scalloped hammerhead shark *Sphyrna lewini*, utilizes deep-water, hypoxic zone in the Gulf of California. *Journal of Fish Biology* 74(7):1682-1687.
- Joung, S. J., N. F. Chen, H. H. Hsu, and K. M. Liu. 2016. Estimates of life history parameters of the oceanic whitetip shark, *Carcharhinus longimanus*, in the Western North Pacific Ocean. *Mar. Biol. Res.*:1-11.
- Júnior, T. V., C. M. Vooren, and R. P. Lessa. 2009. Feeding strategy of the night shark (*Carcharhinus signatus*) and scalloped hammerhead shark (*Sphyrna lewini*) near seamounts off northeastern Brazil. *Brazilian Journal of Oceanography* 57(2):97-104.
- Kahn, J. E., and coauthors. 2014. Atlantic sturgeon annual spawning run estimate in the Pamunkey River, Virginia. *Transactions of the American Fisheries Society* 143(6):1508-1514.
- Kahnle, A. W., K. A. Hattala, and K. A. McKown. 2007. Status of Atlantic sturgeon of the Hudson River Estuary, New York, USA. *American Fisheries Society Symposium* 56:347-363.
- Kahnle, A. W., and coauthors. 1998. Stock Status of Atlantic sturgeon of Atlantic Coast Estuaries. *Atlantic States Marine Fisheries Commission*.
- Keinath, J. A., and J. A. Musick. 1993. Movements and diving behavior of a leatherback turtle, *Dermochelys coriacea*. *Copeia* 1993(4):1010-1017.
- King, T. L., B. A. Lubinski, and A. P. Spidle. 2001. Microsatellite DNA variation in Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) and cross-species amplification in the Acipenseridae. *Conservation Genetics* 2(2):103-119.

- Kiyota, M., K. Yokota, T. Nobetsu, H. Minami, and H. Nakano. 2004. Assessment of mitigation measures to reduce interactions between sea turtles and longline fishery. Proceedings International Symposium SEASTAR2000 Bio-logging Science 2004:24-29.
- Klimley, A. P. 1993. Highly directional swimming by scalloped hammerhead sharks, *Sphyrna lewini*, and subsurface irradiance, temperature, bathymetry, and geomagnetic field. Marine Biology 117(1):1-22.
- Kohler, N. E., and P. A. Turner. 2001. Shark tagging: a review of conventional methods and studies. Pages 191-224 in The behavior and sensory biology of elasmobranch fishes: an anthology in memory of Donald Richard Nelson. Springer.
- Kotas, J. E., V. Mastrochirico, and M. Petrere Junior. 2011. Age and growth of the Scalloped Hammerhead shark, *Sphyrna lewini* (Griffith and Smith, 1834), from the southern Brazilian coast. Brazilian Journal of Biology 71(3):755-761.
- Kotas, J. E., M. Petrere Junior, F. Fiedler, V. Mastrochirico, and G. Sales. 2008. A pesca de emalhe-de-superfície de santa catarina direcionada à captura dos tubarões-martelo, *Sphyrna lewini* (Griffith & Smith 1834) e *Sphyrna zygaena* (Linnaeus 1758). Atlântica, Rio Grande 30:113-128.
- KRRMP. 1993. Kennebec River Resource Management Plan: Balancing Hydropower Generation and Other Uses. Final Report to the Maine State Planning Office, Augusta, ME.
- Kubis, S. A., M. Chaloupka, L. M. Ehrhart, and M. Bresette. 2009. Growth rates of juvenile green turtles *Chelonia mydas* from three ecologically distinct foraging habitats along the east central coast of Florida, USA. Marine Ecology Progress Series 389:257-269.
- Kynard, B., and M. Horgan. 2002. Ontogenetic behavior and migration of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, and shortnose sturgeon, *A. brevirostrum*, with notes on social behavior. Environmental Biology of Fishes 63(2):137-150.
- Kyne, P. M., and coauthors. 2012. The Conservation Status of North American, Central American, and Caribbean Chondrichthyans. IUCN Species Survival Commission Shark Specialist Group., 0956106323.
- LaBrecque, E., C. Curtice, J. Harrison, S. M. Van Parijs, and P. N. Halpin. 2015. 3. Biologically Important Areas for Cetaceans Within U.S. Waters – Gulf of Mexico Region. Aquatic Mammals 41(1):30-38.
- Lagueux, C. J. 2001. Status and distribution of the green turtle, *Chelonia mydas*, in the wider Caribbean region. Pages 32-35 in K. L. Eckert, and F. A. Abreu Grobois, editors. Marine Turtle Conservation in the Wider Caribbean Region - A Dialogue for Effective Regional Management, Santo Domingo, Dominican Republic.

- Lahanas, P. N., and coauthors. 1998. Genetic composition of a green turtle (*Chelonia mydas*) feeding ground population: Evidence for multiple origins. *Marine Biology* 130(3):345-352.
- Laney, R. W., J. E. Hightower, B. R. Versak, M. F. Mangold, and S. E. Winslow. 2007. Distribution, habitat use, and size of Atlantic sturgeon captured during cooperative winter tagging cruises, 1988-2006. *Amercian Fisheries Society Symposium* 56:167-182.
- Laurent, L., and coauthors. 1998. Molecular resolution of marine turtle stock composition in fishery by-catch: A case study in the Mediterranean. *Molecular Ecology* 7:1529-1542.
- Law, R. J., and coauthors. 1991a. Concentrations of Trace-Metals in the Livers of Marine Mammals (Seals, Porpoises and Dolphins) from Waters around the British-Isles. *Marine Pollution Bulletin* 22(4):183-191.
- Law, R. J., and coauthors. 1991b. Concentrations of trace metals in the livers of marine mammals (seals, porpoises and dolphins) from waters around the British Isles. *Marine Pollution Bulletin* 22(4):183-191.
- Leland, J. G. 1968. A survey of the sturgeon fishery of South Carolina. Bears Bluff Laboratories, Wadmalaw Island, S.C.
- León, Y. M., and C. E. Diez. 1999. Population structure of hawksbill turtles on a foraging ground in the Dominican Republic. *Chelonian Conservation and Biology* 3(2):230-236.
- León, Y. M., and C. E. Diez. 2000. Ecology and population biology of hawksbill turtles at a Caribbean feeding ground. Pages 32-33 in F. A. Abreu-Grobois, R. Briseño-Dueñas, R. Márquez-Millán, and L. Sarti-Martinez, editors. Eighteenth International Sea Turtle Symposium. U.S. Department of Commerce, Mazatlán, Sinaloa, México.
- Lessa, R., F. M. Santana, and R. Paglerani. 1999. Age, growth and stock structure of the oceanic whitetip shark, *Carcharhinus longimanus*, from the southwestern equatorial Atlantic. *Fisheries Research* 42:21-30.
- Lezama, C. 2009. impacto de la pesqueria artesanal sobre la tortoga verde (*Chelonia mydas*) en las costas del Rio de la Plata exterior. Universidad de la República.
- Lima, E. H. S. M., M. T. D. Melo, and P. C. R. Barata. 2010. Incidental capture of sea turtles by the lobster fishery off the Ceará Coast, Brazil. *Marine Turtle Newsletter* 128:16-19.
- Limpus, C. J. 1992. The hawksbill turtle, *Eretmochelys imbricata*, in Queensland: Population struture within a southern Great Barrier Reef feeding ground. *Australian Wildlife Research* 19:489-506.

- Limpus, C. J., and J. D. Miller. 2000. Final report for Australian hawksbill turtle population dynamics project. Queensland Parks and Wildlife Service.
- Liu, K. M., and C. T. Chen. 1999. Demographic analysis of the scalloped hammerhead, *Sphyrna lewini*, in the northwestern Pacific. *Fisheries Science* 65(2):218-223.
- Longwell, A., S. Chang, A. Hebert, J. Hughes, and D. Perry. 1992. Pollution and developmental abnormalities of Atlantic fishes. *Environmental Biology of Fishes* 35(1):1-21.
- López-Barrera, E. A., G. O. Longo, and E. L. A. Monteiro-Filho. 2012. Incidental capture of green turtle (*Chelonia mydas*) in gillnets of small-scale fisheries in the Paranaguá Bay, Southern Brazil. *Ocean and Coastal Management* 60:11-18.
- López-Mendilaharsu, M., A. Estrades, M. A. C. Caraccio, V., M. Hernández, and V. Quirici. 2006. Biología, ecología y etología de las tortugas marinas en la zona costera uruguay, Montevideo, Uruguay: Vida Silvestre, Uruguay.
- Lund, F. P. 1985. Hawksbill turtle (*Eretmochelys imbricata*) nesting on the East Coast of Florida. *Journal of Herpetology* 19(1):166-168.
- Lutcavage, M., P. Plotkin, B. Witherington, and P. Lutz. 1997. Human impacts on sea turtle survival. Pages 387–409 in P. Lutz, and J. A. Musick, editors. *The Biology of Sea Turtles*, volume 1. CRC Press, Boca Raton, Florida.
- Lutcavage, M. E., and P. L. Lutz. 1997. Diving physiology. Pages 277-295 in *The Biology of Sea Turtles*. CRC Press, Boca Raton, Florida.
- Mac, M. J., and C. C. Edsall. 1991. Environmental contaminants and the reproductive success of Lake Trout in the Great Lakes: an epidemiological approach. *Journal of Toxicology and Environmental Health* 33:375-394.
- Mackay, A. L. 2006. 2005 sea turtle monitoring program the East End beaches (Jack's, Isaac's, and East End Bay) St. Croix, U.S. Virgin Islands. Nature Conservancy.
- MAFMC. 2007a. 2008 Atlantic mackerel, squid, and butterfish specifications, including an Environmental Assessment, Regulatory Impact Review, and Initial Flexibility Analysis. Mid-Atlantic Fishery Management Council. November 2007.
- MAFMC. 2007b. 2008 Summer Flounder, Scup, and Black Sea Bass Specifications Environmental Assessment Regulatory Impact Review Initial Regulatory Flexibility Analysis and Essential Fish Habitat Assessment Mid-Atlantic Fishery Management Council Dover, Delaware.
- MAFMC. 2010. Spiny Dogfish Specifications, Environmental Assessment, Regulatory Impact Review, and Initial Regulatory Flexibility Analysis. Mid-Atlantic Fishery Management Council

- MAFMC. 2013. 2013 and 2014 Bluefish Specifications, Environmental Assessment, and Initial Regulatory Flexibility Analysis. Mid-Atlantic Fishery Management Council. April 2013.
- MAFMC and ASMFC. 1998. Amendment 1 to the Bluefish Fishery Management Plan with a Supplemental Environmental Impact Statement and Regulatory Impact Review. Mid-Atlantic Fishery Management Council. October 1998. Mid-Atlantic Fishery Management Council and Atlantic States Marine Fisheries Commission
- Maguire, J.-J., M. Sissenwine, J. Csirke, and R. Grainger. 2006. The state of the world highly migratory, straddling and other high seas fish stocks, and associated species. United Nations, Food and Agriculture Organization, Rome, Italy.
- Maharaj, A. M. 2004. A comparative study of the nesting ecology of the leatherback turtle *Dermochelys coriacea* in Florida and Trinidad. University of Central Florida, Orlando, Florida.
- Marcovaldi, N., B. B. Gifforni, H. Becker, F. N. Fiedler, and G. Sales. 2009. Sea Turtle Interactions in Coastal Net Fisheries in Brazil. U.S. National Marine Fisheries Service, Southeast Fisheries Science Center: Honolulu, Gland, Switze, Honolulu, Hawaii, USA.
- Marcy, B. C., D. E. Fletcher, F. D. Martin, M. H. Paller, and J. M. Reichert. 2005. Fishes of the middle Savannah River Basin: with emphasis on the Savannah River Site. University of Georgia Press.
- Márquez M., R. 1990. Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date, Rome.
- Márquez M., R. 1994. Synopsis of biological data on the Kemp's ridley sea turtle, *Lepidochelys kempii* (Garman, 1880). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Center.
- Matkin, C. O., and E. Saulitis. 1997. Restoration notebook: Killer whale (*Orcinus orca*). Exxon Valdez Oil Spill Trustee Council, Anchorage, Alaska.
- Matos, R. 1986. Sea turtle hatchery project with specific reference to the leatherback turtle (*Dermochelys coriacea*), Humacao, Puerto Rico 1986. Puerto Rico Department of Natural Resources, de Tierra, Puerto Rico.
- Matta, M. B., C. Cairncross, and R. M. Kocan. 1997. Effect of a polychlorinated biphenyl metabolite on early life stage survival of two species of trout. Bulletin of Environmental Contamination and Toxicology 59:146-151.
- Mayor, P. A., B. Phillips, and Z.-M. Hillis-Starr. 1998. Results of the stomach content analysis on the juvenile hawksbill turtles of Buck Island Reef National Monument, U.S.V.I. Pages 230-233 in S. P. Epperly, and J. Braun, editors. Seventeenth Annual Sea Turtle Symposium.

- McCord, J. W., M. R. Collins, W. C. Post, and T. I. J. Smith. 2007. Attempts to develop an index of abundance for age-1 Atlantic sturgeon in South Carolina, USA. *American Fisheries Society Symposium* 56:397-403.
- McDonald, D. L., and P. H. Dutton. 1996. Use of PIT tags and photoidentification to revise remigration estimates of leatherback turtles (*Dermochelys coriacea*) nesting in St. Croix, U.S. Virgin Islands, 1979-1995. *Chelonian Conservation and Biology* 2(2):148-152.
- McKenzie, C., B. J. Godley, R. W. Furness, and D. E. Wells. 1999. Concentrations and patterns of organochlorine contaminants in marine turtles from Mediterranean and Atlantic waters. *Marine Environmental Research* 47:117-135.
- McMichael, E., R. R. Carthy, and J. A. Seminoff. 2003. Evidence of homing behavior in juvenile green turtles in the northeastern Gulf of Mexico. Pages 223-224 in J. A. Seminoff, editor *Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation*.
- Meehl, G. A., and coauthors. 2007. Global climate projections. Pages 747-846 in S. Solomon, and coeditors, editors. *Climate Change 2007: The Physical Science Basis*. Cambridge University Press, Cambridge, UK and New York, NY.
- Meylan, A., B. 1988. Spongivory in hawksbill turtles: A diet of glass. *Science* 239(4838):393-395.
- Meylan, A., B., B. Schroeder, and A. Mosier. 1994. Marine turtle nesting activity in the State of Florida, 1979-1992. Pages 83 in K. A. Bjorndal, A. B. Bolten, D. A. Johnson, and P. J. Eliazar, editors. *Fourteenth Annual Symposium on Sea Turtle Biology and Conservation*.
- Meylan, A., and M. Donnelly. 1999. Status justification for listing the hawksbill turtle (*Eretmochelys imbricata*) as critically endangered on the 1996 IUCN Red List of threatened animals. *Chelonian Conservation and Biology* 3(2):200-224.
- Meylan, A. B. 1999a. International movements of immature and adult hawksbill turtles (*Eretmochelys imbricata*) in the Caribbean region. *Chelonian Conservation and Biology* 3(2):189-194.
- Meylan, A. B. 1999b. Status of the hawksbill turtle (*Eretmochelys imbricata*) in the Caribbean region. *Chelonian Conservation and Biology* 3(2):177-184.
- Meylan, A. B., B. A. Schroeder, and A. Mosier. 1995. Sea turtle nesting activity in the State of Florida 1979-1992. *Florida Department of Environmental Protection* (52):63.
- Meylan, A. B., B. E. Witherington, B. Brost, R. Rivera, and P. S. Kubilis. 2006. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of *Caretta*, *Chelonia*, and *Dermochelys*. Pages 306-307 in

- M. Frick, A. Panagopoulou, A. F. Rees, and K. Williams, editors. 26th Annual Symposium on Sea Turtle Biology and Conservation, 3-8 April 2006, Island of Crete, Greece, Book of Abstracts. International Sea Turtle Society, Athens, Greece.
- Meylan, P. A., A. B. Meylan, and J. A. Gray. 2011. The ecology and migrations of sea turtles 8. Tests of the developmental habitat hypothesis. *Bulletin of the American Museum of Natural History* 357:1-70.
- Miller, J. D. 1997. Reproduction in sea turtles. Pages 51-58 *in* P. L. Lutz, and J. A. Musick, editors. *The Biology of Sea Turtles*. CRC Press, Boca Raton, Florida.
- Miller, M. H., and coauthors. 2014. Status Review Report: Scalloped Hammerhead Shark (*Sphyrna lewini*). National Marine Fisheries Service - Office of Protected Resources.
- Miller, T., and G. Shepherd. 2011. Summary of Discard Estimates for Atlantic Sturgeon. Population Dynamics Branch, Northeast Fisheries Science Center.
- Milliken, H. O., and coauthors. 2007. Evaluation of a modified scallop dredge's ability to reduce the likelihood of damage to loggerhead sea turtle carcasses. NOAA, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, Massachusetts.
- Milliken, T., and H. Tokunaga. 1987. The Japanese sea turtle trade 1970-1986. TRAFFIC (JAPAN), Center for Environmental Education, Washington, D. C.
- Milton, S. L., and P. L. Lutz. 2003. Physiological and genetic responses to environmental stress. Pages 163-197 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. *The Biology of Sea Turtles*, volume II. CRC Press, Boca Raton, Florida.
- Mo, C. L. 1988. Effect of bacterial and fungal infection on hatching success of Olive Ridley sea turtle eggs. World Wildlife Fund-U.S.
- Moncada, F., and coauthors. 2010. Movement patterns of loggerhead turtles *Caretta caretta* in Cuban waters inferred from flipper tag recaptures. *Endangered Species Research* 11(1):61-68.
- Moncada, F., and coauthors. 2006. Movement patterns of green turtles (*Chelonia mydas*) in Cuba and adjacent Caribbean waters inferred from flipper tag recaptures. *Journal of Herpetology* 40(1):22-34.
- Moncada, F., E. Carrillo, A. Saenz, and G. Nodarse. 1999. Reproduction and nesting of the hawksbill turtle, *Eretmochelys imbricata*, in the Cuban Archipelago. *Chelonian Conservation and Biology* 3(2):257-263.
- Moncada Gavilan, F. 2001. Status and distribution of the loggerhead turtle, *Caretta caretta*, in the wider Caribbean region. Pages 36-40 *in* K. L. Eckert, and F. A. Abreu Grobois, editors. *Marine Turtle Conservation in the Wider Caribbean Region*

- A Dialogue for Effective Regional Management, Santo Domingo, Dominican Republic.

- Monzón-Argüello, C., and coauthors. 2010. Evidence from genetic and Lagrangian drifter data for transatlantic transport of small juvenile green turtles. *Journal of Biogeography* 37(9):1752-1766.
- Moore, A., and C. P. Waring. 2001. The effects of a synthetic pyrethroid pesticide on some aspects of reproduction in Atlantic salmon (*Salmo salar* L.). *Aquatic Toxicology* 52(1):1-12.
- Morgan, A., P. W. Cooper, T. Curtis, and G. H. Burgess. 2009. Overview of the U.S. East Coast Bottom Longline Shark Fishery, 1994–2003. . *Marine Fisheries Review* 71(1):23-38.
- Mortimer, J. A., and A. Carr. 1987. Reproduction and migrations of the Ascension Island green turtle (*Chelonia mydas*). *Copeia* 1987(1):103-113.
- Mortimer, J. A., and coauthors. 2003. Growth rates of immature hawksbills (*Eretmochelys imbricata*) at Aldabra Atoll, Seychelles (Western Indian Ocean). Pages 247-248 in J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation.
- Mortimer, J. A., M. Day, and D. Broderick. 2002. Sea turtle populations of the Chagos Archipelago, British Indian Ocean Territory. Pages 47-49 in A. Mosier, A. Foley, and B. Brost, editors. Twentieth Annual Symposium on Sea Turtle Biology and Conservation.
- Mortimer, J. A., and M. Donnelly. 2008. Hawksbill turtle (*Eretmochelys imbricata*) International Union for Conservation of Nature and Natural Resources.
- Moser, M. L., J. B. Bichy, and S. B. Roberts. 1998. Sturgeon Distribution in North Carolina. Center for Marine Science Research, Wilmington, North Carolina.
- Moser, M. L., and S. W. Ross. 1995. Habitat use and movements of shortnose and Atlantic sturgeons in the lower Cape Fear River, North Carolina. *Transactions of the American Fisheries Society* 124(2):225.
- Moyes, C. D., N. Fragoso, M. K. Musyl, and R. W. Brill. 2006. Predicting postrelease survival in large pelagic fish. *Transactions of the American Fisheries Society* 135(5):1389-1397.
- Mrosovsky, N., G. D. Ryan, and M. C. James. 2009. Leatherback turtles: The menace of plastic. *Marine Pollution Bulletin* 58(2):287-289.
- Munro, J., R. E. Edwards, and A. W. Kahnle. 2007. Anadromous Sturgeons: Habitats, Threats, and Management Synthesis and Summary. *American Fisheries Society Symposium* 56:1-15.

- Murawski, S. A., A. L. Pacheco, and United States. National Marine Fisheries Service. 1977. Biological and fisheries data on Atlantic sturgeon, *Acipenser oxyrinchus* (Mitchill). Sandy Hook Laboratory, Northeast Fisheries Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Dept. of Commerce, Highlands, N.J.
- Murdoch, P. S., J. S. Baron, and T. L. Miller. 2000. Potential Effects of Climate Change of Surface Water Quality in North America. *JAWRA Journal of the American Water Resources Association* 36(2):347-366.
- Murphy, T. M., and S. R. Hopkins. 1984. Aerial and ground surveys of marine turtle nesting beaches in the southeast region. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center.
- Murray, K. T. 2009a. Characteristics and magnitude of sea turtle bycatch in US mid-Atlantic gillnet gear. *Endangered Species Research* 8(3):211-224.
- Murray, K. T. 2009b. Proration of estimated bycatch of loggerhead sea turtles in U.S. mid-Atlantic sink gillnet gear to vessel trip report landed catch, 2002-2006. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.
- Murray, K. T. 2011. Interactions between seaturtles and dredge gear in the U.S. sea scallop (*Placopecten magellanicus*) fishery, 2001–2008. *Fisheries Research* 107(1-3):137-146.
- Musick, J. A. 1999. Life in the Slow Lane: Ecology and Conservation of Long-Lived Marine Animals. Pages 1-10 *in* Symposium Conservation of Long-Lived Marine Animals. American Fisheries Society, Monterey, California, USA
- Musick, J. A., and C. J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pages 137-163 *in* P. L. Lutz, and J. A. Musick, editors. *The Biology of Sea Turtles*. CRC Press, New York, New York.
- Musyl, M. K., and coauthors. 2011. Postrelease survival, vertical and horizontal movements, and thermal habitats of five species of pelagic sharks in the central Pacific Ocean. *Fishery Bulletin* 109:341- 368.
- Naro-Maciel, E., J. H. Becker, E. H. S. M. Lima, M. A. Marcovaldi, and R. DeSalle. 2007. Testing dispersal hypotheses in foraging green sea turtles (*Chelonia mydas*) of Brazil. *Journal of Heredity* 98(1):29-39.
- Naro-Maciel, E., and coauthors. 2012. The interplay of homing and dispersal in green turtles: A focus on the southwestern atlantic. *Journal of Heredity* 103(6):792-805.
- NAST. 2000. Climate change impacts on the United States: the potential consequences of climate variability and change. US Global Change Research Program, Washington D.C. National Assessment Synthesis Team.

- NEFMC. 1982 Fishery Management Plan, Final Environmental Impact Statement, Regulatory Impact Review for Atlantic sea scallops (*Placopecten magellanicus*). Prepared by the New England Fishery Management Council in consultation with Mid-Atlantic Fishery Management Council and South Atlantic Fishery Management Council. January 1982.
- NEFMC. 2003. Final Amendment 10 to the Atlantic Sea Scallop Fishery Management Plan with a Supplemental Environmental Impact Statement, Regulatory Impact Review, and Regulatory Flexibility Analysis. New England Fishery Management Council. November 2003.
- NEFMC. 2009. Amendment 16 to the Northeast Multispecies Fishery Management Plan including a Final Supplemental Environmental Impact Statement and Initial Regulatory Flexibility Analysis. October 16, 2009.
- NEFSC. 2003. Assessment of Spiny Dogfish. Pages 133-283 in 37th Northeast Regional Stock Assessment Workshop (37th SAW). NEFSC Reference Document 03-16. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, Massachusetts.
- NEFSC. 2005. 40th Northeast Regional Stock Assessment Workshop (40th SAW). NEFSC Reference Document 05-04. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, Massachusetts.
- NEFSC. 2006. 43rd Northeast Regional Stock Assessment Workshop (43rd SAW), Stock Assessment Review Committee (SARC) consensus summary of assessments. NEFSC Reference Document 06-25. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, Massachusetts.
- NEFSC. 2007a. Assessment of Atlantic Sea Scallops. Pages 139-370 in 45th Northeast Regional Stock Assessment Workshop (45th SAW). NEFSC Reference Document 07-16. 380 pp.
- NEFSC. 2007b. Assessment of Northeast Skate Species Complex. Pages 285-547 in 44th Northeast Regional Stock Assessment Workshop (44th SAW). National Marine Fisheries Service, Woods Hole, MA.
- Niklitschek, E. J., and D. H. Secor. 2005. Modeling spatial and temporal variation of suitable nursery habitats for Atlantic sturgeon in the Chesapeake Bay. *Estuarine, Coastal and Shelf Science* 64(1):135-148.
- Niklitschek, E. J., and D. H. Secor. 2009a. Dissolved oxygen, temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: I. Laboratory results. *Journal of Experimental Marine Biology and Ecology* 381(Supplement 1):S150-S160.

- Niklitschek, E. J., and D. H. Secor. 2009b. Dissolved oxygen, temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: II. Model development and testing. *Journal of Experimental Marine Biology and Ecology* 381(Supplement 1):S161-S172.
- Niklitschek, E. J., and D. H. Secor. 2010. Experimental and field evidence of behavioural habitat selection by juvenile Atlantic *Acipenser oxyrinchus oxyrinchus* and shortnose *Acipenser brevirostrum* sturgeons. *Journal of Fish Biology* 77(6):1293-1308.
- NMFS-SEFSC. 2009. An assessment of loggerhead sea turtles to estimate impacts of mortality on population dynamics. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, PRD-08/09-14.
- NMFS. 1997a. Biological Opinion on Navy activities off the southeastern United States along the Atlantic Coast. Submitted on May 15, 1997.
- NMFS. 1997b. Endangered Species Act Section 7 Consultation - Biological Opinion on the continued hopper dredging of channels and borrow areas in the southeastern United States. Submitted on September 25, 1997.
- NMFS. 1998. Final recovery plan for the shortnose sturgeon (*Acipenser brevirostrum*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, Maryland.
- NMFS. 2000. Smalltooth Sawfish Status Review. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.
- NMFS. 2001a. Biological Opinion on the Reinitiation of Consultation on the Atlantic Highly Migratory Species Fishery Management Plan and its Associated Fisheries. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Endangered Species Division, Silver Spring, Maryland.
- NMFS. 2001b. Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the western North Atlantic. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.
- NMFS. 2002a. Biological Opinion on Shrimp Trawling in the Southeastern United States, under the Sea Turtle Conservation Regulations and as Managed by the Fishery Management Plans for Shrimp in the South Atlantic and Gulf of Mexico. Submitted on December 2, 2002. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, Florida.

- NMFS. 2002b. Biological Opinion on the Implementation of the Deep-Sea Red Crab, *Chaceon quinquefrenatus*, Fishery Management Plan, St. Petersburg, Florida.
- NMFS. 2002c. Endangered Species Act - Section 7 consultation, biological opinion. Shrimp trawling in the southeastern United States under the sea turtle conservation regulations and as managed by the fishery management plans for shrimp in the South Atlantic and Gulf of Mexico. National Marine Fisheries Service, Southeast Regional Office, St. Petersburg, Florida.
- NMFS. 2003a. Biological Opinion on the continued operation of Atlantic shark fisheries (commercial shark bottom longline and drift gillnet fisheries and recreational shark fisheries) under the fishery management plan for Atlantic tunas, swordfish, and sharks (HMS FMP) and the proposed rule for draft amendment 1 to the HMS FMP. Submitted on July 2003. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, St. Petersburg, Florida.
- NMFS. 2003b. Biological Opinion on the review of January 2003 FMP for 2003 Dolphin and Wahoo Fishery of the Atlantic. Submitted on August 27, 2003 National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2002-1305, St. Petersburg, Florida.
- NMFS. 2004. Biological Opinion on Reinitiation of Consultation on Atlantic Pelagic Longline Fishery for Highly Migratory Species. Submitted on June 1, 2004. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2004-80, Saint Petersburg, Florida.
- NMFS. 2005a. Biological Opinion on the continued authorization of shrimp trawling as managed under the Fishery Management Plan (FMP) for the shrimp fishery of the South Atlantic region, including proposed Amendment 6 to that FMP.
- NMFS. 2005b. Endangered Species Act Section 7 Consultation - Review of Amendment 2 to the Monkfish Fishery Management Plan. Submitted on April 1, 2005. National Marine Fisheries Service NER-2004-2230, Gloucester, Massachusetts.
- NMFS. 2006. Final Consolidated Atlantic Highly Migratory Species Fishery Management Plan. Pages 1629 *in*.
- NMFS. 2007. Biological Opinion on the reinitiation of consultation on Atlantic Pelagic Longline Fishery for Highly Migratory Species. Submitted on March 2, 2007. National Marine Fisheries Service, SER-2004-2590, St. Petersburg, Florida.
- NMFS. 2008a. Biological Opinion on the Atlantic Sea Scallop Fishery Management Plan. March 14, 2008.
- NMFS. 2008b. Biological Opinion on the Continued Authorization of Shark Fisheries (Commercial Shark Bottom Longline, Commercial Shark Gillnet and Recreational

Shark Handgear Fisheries) as Managed under the Consolidated Fishery Management Plan for Atlantic Tunas, Swordfish, and Sharks (Consolidated HMS FMP), including Amendment 2 to the Consolidated HMS FMP. Biological Opinion.

NMFS. 2008c. Endangered Species Act Section 7 Consultation - Biological Opinion on the Continued Authorization of Shark Fisheries (Commercial Shark Bottom Longline, Commercial Shark Gillnet and Recreational Shark Handgear Fisheries) as Managed under the Consolidated Fishery Management Plan for Atlantic Tunas, Swordfish, and Sharks (Consolidated HMS FMP), including Amendment 2 to the Consolidated HMS FMP.

NMFS. 2009. Smalltooth Sawfish Recovery Plan, Silver Spring, MD.

NMFS. 2010. Smalltooth Sawfish 5-Year Review: Summary and Evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Protected Resources Division, St. Petersburg, FL.

NMFS. 2011a. Biological Opinion on Deepening of the Savannah Harbor Federal Navigational Channel in association with the Savannah Harbor Expansion Project

NMFS. 2011b. Final environmental assessment, regulatory impact review, and final regulatory flexibility analysis for a final rule to implement the 2010 International Commission for the Conservation of Atlantic Tunas recommendations on Sharks. United States Department of Commerce National Oceanic and Atmospheric Administration National Marine Fisheries Service Office of Sustainable Fisheries Highly Migratory Species Management Division.

NMFS. 2011c. Final recovery plan for the sei whale (*Balaenoptera borealis*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS. 2012a. 2012 Stock Assessment and Fishery Evaluation report (SAFE) for Atlantic Highly Migratory Species. Atlantic Highly Migratory Species Management Division. NOAA Fisheries. U.S. Department of Commerce.

NMFS. 2012b. Biological Opinion on Reinitiation of Endangered Species Act Section 7 Consultation on the Continued Implementation of the Sea Turtle Conservation Regulations, as Proposed to Be Amended, and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, Florida.

NMFS. 2012c. Biological Opinion on the Continued Authorization of the Atlantic Shark Fisheries via the Consolidated HMS Fishery Management Plan as Amended by Amendments 3 and 4 and the Federal Authorization of a Smoothhound Fishery.

- NMFS. 2012d. Protocols for Categorizing Sea Turtles for Post-release Mortality Estimates. August 2001, revised February 2012. PRD Contribution: #PRD-2011-07. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, Florida.
- NMFS. 2012e. Reinitiation of Endangered Species Act (ESA) Section 7 Consultation on the Continued Implementation of the Sea Turtle Conservation Regulations, as Proposed to Be Amended, and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Act. Biological Opinion. NOAA, NMFS, SERO, Protected Resources Division (F/SER3) and Sustainable Fisheries Division (F/SER2).
- NMFS. 2013a. Biological Opinion 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. NER-2012-01956], Submitted on December 16, 2013. National Marine Fisheries Service, Northeast Regional Office, Protected Resources Division.
- NMFS. 2013b. Hawksbill sea turtle (*Eremochelys imbricata*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service.
- NMFS. 2013c. Permit 16645 for Take of Listed Sturgeon Incidental to the Georgia Commercial Shad Fishery. National Marine Fisheries Service, Silver Spring, MD.
- NMFS. 2014a. 2014 Stock Assessment and Fishery Evaluation (SAFE) Report for Atlantic Highly Migratory Species. Atlantic Highly Migratory Species Management Division. NOAA Fisheries. U.S. Department of Commerce, 195 pp.
- NMFS. 2014b. Biological Opinion on the Continued Implementation of the Sea Turtle Conservation Regulations under the ESA and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, Florida.
- NMFS. 2014c. Biological Opinion on the Continued Implementation of the Sea Turtle Conservation Regulations under the ESA and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act (MSFMCA), St. Petersburg, Florida.
- NMFS. 2014d. Biological Opinion on the Office of Protected Resources' proposal to issue an Incidental Take Permit for Atlantic sturgeon affected by North Carolina's inshore anchored gill net fishery pursuant to section 10(a)(1)(B) of the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Spring, MD.

- NMFS. 2014e. Final Amendment 7 to the 2006 Consolidated Atlantic Highly Migratory Species Fishery Management Plan. Pages 796 *in*.
- NMFS. 2014f. Request for reinitiation of formal consultation on the 2001 biological opinion on the Atlantic Highly Migratory Species Fishery Management Plan and the 2012 Shark and Smoothhound biological opinion to assess effects to the scalloped hammerhead shark Central and Southwest Atlantic Distinct Population Segment and seven species of Coral.
- NMFS. 2015a. Biological Evaluation on the Effects of HMS Fishery Interactions with the Central and Southwest Atlantic DPS of Scalloped Hammerhead Shark (*Sphyrna lewini*) and Seven Threatened Coral Species. Draft Report from the Office of Sustainable Fisheries, revised July 2015., Silver Spring, Maryland.
- NMFS. 2015b. Biological Opinion on the Reinitiation of the Continued Authorization of the Fishery Management Plan for Coastal Migratory Pelagic Resources in the Atlantic and Gulf of Mexico under the Magnuson-Stevens Fishery Management and Conservation Act. Submitted on June 18, 2015. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division (F/SER3), and Sustainable Fisheries Division (F/SER2), St. Petersburg, Florida.
- NMFS. 2016. Endangered Species Act Section 7 Consultation on the Continued Prosecution of Fisheries and Ecosystem Research Conducted and Funded by the Northeast Fisheries Science Center and the Issuance of a Letter of Authorization under the Marine Mammal Protection Act for the Incidental Take of Marine Mammals Pursuant to those Research Activities. PCTS ID: NER-2015-12532. June 23, 2016.
- NMFS. 2017a. Endangered Species Act (ESA) Section 7 Consultation on United States Fish and Wildlife Service (USFWS) Funding of Georgia Department of Natural Resources (GA DNR) to Collect, Analyze and Report Biological and Fisheries Information to Describe the Conditions or Health of Recreationally Important Finfish (SER 20 15-16739). N. NOAA, SERO, Protected Resources Division (F/SER3), editor.
- NMFS. 2017b. National Marine Fisheries Service Procedural Instruction 02-110-21. Process for Post-Interaction Mortality Determinations Of Sea Turtles Bycaught In Trawl, Net, and Pot/Trap Fisheries. .
- NMFS, USFWS, and SEMARNAT. 2011a. Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. Pages 156 *in*. National Marine Fisheries Service, Silver Spring, Maryland.
- NMFS, USFWS, and SEMARNAT. 2011b. Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. National Marine Fisheries Service, Silver Spring, Maryland.

- NMFS, USFWS, and SEMARNAT. 2011c. BiNational Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. National Marine Fisheries Service, Silver Spring, Maryland.
- NMFS, USFWS, and SEMARNAT. 2011d. BiNational Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. . Pages 156 *in*.
- NMFS and USFWS. 1991. Recovery plan for U.S. Population of Atlantic Green Turtle (*Chelonia mydas*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.
- NMFS and USFWS. 1992. Recovery plan for leatherback turtles *Dermochelys coriacea* in the U. S. Caribbean, Atlantic and Gulf of Mexico. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NMFS and USFWS. 1993. Recovery plan for the hawksbill turtle *Eretmochelys imbricata* in the U.S. Caribbean, Atlantic and Gulf of Mexico. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, St. Petersburg, Florida.
- NMFS and USFWS. 1995. Status reviews for sea turtles listed under the Endangered Species Act of 1973. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.
- NMFS and USFWS. 1998a. Recovery plan for U. S. Pacific populations of the hawksbill turtle (*Eretmochelys imbricata*). National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NMFS and USFWS. 1998b. Recovery plan for U.S. Pacific populations of the leatherback turtle (*Dermochelys coriacea*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.
- NMFS and USFWS. 2007a. Green Sea Turtle (*Chelonia mydas*) 5-year review: Summary and Evaluation. National Marine Fisheries Service, Silver Spring, Maryland.
- NMFS and USFWS. 2007b. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NMFS and USFWS. 2007c. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NMFS and USFWS. 2007d. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

- NMFS and USFWS. 2007e. Loggerhead sea turtle (*Caretta caretta*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NMFS and USFWS. 2008. Recovery plan for the northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*), second revision. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.
- NMFS and USFWS. 2009. Recovery plan for the northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*). National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NMFS and USFWS. 2013. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. NOAA, National Marine Fisheries Service, Office of Protected Resources and U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Office.
- NMFS and USFWS. 2015. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: summary and evaluation. National Marine Fisheries Service, U.S. Fish and Wildlife Service, Silver Spring, Maryland.
- NOAA. 2012. Understanding Climate. <http://www.climate.gov/#understandingClimate>.
- Noriega, R., J. M. Werry, W. Sumpton, D. Mayer, and S. Y. Lee. 2011. Trends in annual CPUE and evidence of sex and size segregation of *Sphyrna lewini*: Management implications in coastal waters of northeastern Australia. Fisheries Research 110(3):472-477.
- Norman, J. R., and F. C. Fraser. 1937. Giant fishes, whales and dolphins. Putman and Company, Limited, London.
- NRC. 1990. Decline of the sea turtles: Causes and prevention. National Research Council, Washington, D. C.
- Ogren, L. H. 1989a. Distribution of juvenile and subadult Kemp's ridley sea turtles: Preliminary results from 1984-1987 surveys. Pages 116-123 in C. W. Caillouet Jr., and A. M. Landry Jr., editors. First International Symposium on Kemp's Ridley Sea Turtle Biology, Conservation and Management. Texas A&M University, Sea Grant College, Galveston, Texas.
- Ogren, L. H. 1989b. Status report of the green turtle. Pages 89-94 in L. Ogren, and coeditors, editors. Second Western Atlantic Turtle Symposium.
- Orlando, S. P., Jr. , and coauthors. 1994. Salinity Characteristics of South Atlantic Estuaries. NOAA, Office of Ocean Resources Conservation and Assessment, Silver Spring, MD.

- Paladino, F. V., M. P. O'Connor, and J. R. Spotila. 1990. Metabolism of leatherback turtles, gigantothermy, and thermoregulation of dinosaurs. *Nature* 344:858-860.
- Parsons, J. J. 1972. The hawksbill turtle and the tortoise shell trade. Pages 45-60 in *Études de Géographie Tropicale Offertes a Pierre Gourou*. Mouton, Paris, France.
- Patricio, A. R., X. Velez-Zuazo, C. E. Diez, R. Van Dam, and A. M. Sabat. 2011. Survival probability of immature green turtles in two foraging grounds at Culebra, Puerto Rico. *Marine Ecology Progress Series* 440:217-227.
- Pennington, M. 1983. Efficient estimators of abundance, for fish and plankton surveys. *Biometrics*:281-286.
- Pershing, A. J., and coauthors. 2015. Slow adaptation in the face of rapid warming leads to collapse of the Gulf of Maine cod fishery. *Science* 350(6262):809-812.
- Pfeffer, W. T., J. T. Harper, and S. O'Neel. 2008. Kinematic Constraints on Glacier Contributions to 21st-Century Sea-Level Rise. *Science* 321(5894):1340-1343.
- Piercy, A. N., J. K. Carlson, J. A. Sulikowski, and G. H. Burgess. 2007. Age and growth of the scalloped hammerhead shark, *Sphyrna lewini*, in the north-west Atlantic Ocean and Gulf of Mexico. *Marine and Freshwater Research* 58(1):34-40.
- Pike, D. A., R. L. Antworth, and J. C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the loggerhead seaturtle, *Caretta caretta*. *Journal of Herpetology* 40(1):91-94.
- Plotkin, P., and A. F. Amos. 1990. Effects of anthropogenic debris on sea turtles in the northwestern Gulf of Mexico. Pages 736-743 in R. S. Shoumura, and M. L. Godfrey, editors. *Proceedings of the Second International Conference on Marine Debris*. NOAA Technical Memorandum NMFS SWFSC-154. U.S. Department of Commerce, Honolulu, Hawaii.
- Plotkin, P. T. 2003. Adult migrations and habitat use. Pages 225-241 in P. L. Lutz, J. A. Musick, and J. Wyneken, editors. *The Biology of Sea Turtles*, volume 2. CRC Press.
- Plotkin, P. T., and A. F. Amos. 1988. Entanglement in and ingestion of marine debris by sea turtles stranded along the South Texas coast. Pages 7 in *Supplemental Deliverables under Entanglement-Debris Task No. 3. Debris, Entanglement and Possible Causes of Death in Stranded Sea Turtles (FY88)*.
- Polovina, J. J., D. R. Kobayashi, D. M. Parker, M. P. Seki, and G. H. Balazs. 2000. Turtles on the edge: movement of loggerhead turtles (*Caretta caretta*) along oceanic fronts, spanning longline fishing grounds in the central North Pacific, 1997-1998. *Fisheries Oceanography* 9(1):71-82.

- Post, G. W. 1987. Revised and Expanded Textbook of Fish Health. T.F.H. Publications, New Jersey.
- Poulakis, G. R. 2012. Distribution, Habitat Use, and Movements of Juvenile Smalltooth Sawfish, *Pristis pectinata*, in the Charlotte Harbor Estuarine System, Florida. Florida Institute of Technology, Melbourne, FL.
- Poulakis, G. R., and J. C. Seitz. 2004a. Recent occurrence of the smalltooth sawfish, *Pristis pectinata* (Elasmobranchiomorphi: Pristidae), in Florida Bay and the Florida Keys, with comments on sawfish ecology. Florida Scientist 67(27):27-35.
- Poulakis, G. R., and J. C. Seitz. 2004b. Recent occurrence of the smalltooth sawfish, *Pristis pectinata* (Elasmobranchiomorphi: Pristidae), in Florida Bay and the Florida Keys, with comments on sawfish ecology. Florida Scientist 67(1):27-35.
- Poulakis, G. R., P. W. Stevens, A. A. Timmers, T. R. Wiley, and C. A. Simpfendorfer. 2011. Abiotic affinities and spatiotemporal distribution of the endangered smalltooth sawfish, *Pristis pectinata*, in a south-western Florida nursery. Marine and Freshwater Research 62(10):1165-1177.
- Prince, E. D., M. Ortiz, and A. Venizelos. 2002. A comparison of circle hook and “J” hook performance in recreational catch and release fisheries for billfish. Am. Fish. Soc. Symp. 30:60–79.
- Pritchard, P. C. H. 1969. The survival status of ridley sea-turtles in America. Biological Conservation 2(1):13-17.
- Pritchard, P. C. H., and coauthors. 1983. Manual of sea turtle research and conservation techniques, Second ed. Center for Environmental Education, Washington, D. C.
- Pritchard, P. C. H., and P. Trebbau. 1984. The turtles of Venezuela. SSAR.
- Prosdocimi, L., V. González Carman, D. A. Albareda, and M. I. Remis. 2012. Genetic composition of green turtle feeding grounds in coastal waters of Argentina based on mitochondrial DNA. Journal of Experimental Marine Biology and Ecology 412:37-45.
- Pyzik, L., J. Caddick, and P. Marx. 2004. Chesapeake Bay: Introduction to an ecosystem. EPA 903-R-04-003, CBP/TRS 232100.
- .
- Radakov, D. V., A. D. Motchek, Y. N. Sbikin, R. C. Madruga, and A. S. Lee. 1975. Acerca de la longitud de los peces comerciales en capturas de la zona noroccidental de Cuba. ACADEMIA DE CIENCIAS DE CUBA INSTITUTO DE OCEANOLOGIA, Habana. Cuba.

- Randall, J. E. 1965. Food habits of the Nassau grouper (*Epinephelus striatus*). Pages 13-16 in Association of Island Marine Laboratories of the Caribbean, 6th Meeting, Isla Margarita, Venezuela.
- Rebel, T. P. 1974. Sea Turtles and the Turtle Industry of the West Indies, Florida and the Gulf of Mexico. University of Miami Press, Coral Gables, Florida.
- Reddering, J. S. V. 1988. Prediction of the effects of reduced river discharge on estuaries of the south-eastern Cape Province, South Africa. *South African Journal of Science* 84:726-730.
- Rhodin, A. G. J. 1985. Comparative chondro-osseous development and growth in marine turtles. *Copeia* 1985:752-771.
- Rice, J., and S. Harley. 2012. Stock assessment of oceanic whitetip sharks in the western and central Pacific Ocean. Western and Central Pacific Fisheries Commission Scientific Committee Eighth Regular Session. WCPFC-SC8-2012/SA-WP-06 Rev 1., 53. Pages 53 in.
- Richards, P. M., and coauthors. 2011. Sea turtle population estimates incorporating uncertainty: A new approach applied to western North Atlantic loggerheads *Caretta caretta*. *Endangered Species Research* 15(2):151-158.
- Richardson, B., and D. Secor. 2016. Assessment of Critical Habitats for Recovering the Chesapeake Bay Atlantic Sturgeon Distinct Population Segment. Maryland Department of Natural Resources, Stevensville, MD.
- Richardson, B., and D. Secor. 2017. Assess threats to the reproduction by Atlantic sturgeon through studies on spawning habitats of Chesapeake Bay DPS sturgeon in the Nanticoke estuary. Maryland Department of Natural Resources.
- Richardson, J. I., R. Bell, and T. H. Richardson. 1999. Population ecology and demographic implications drawn from an 11-year study of nesting hawksbill turtles, *Eretmochelys imbricata*, at Jumby Bay, Long Island, Antigua, West Indies. *Chelonian Conservation and Biology* 3(2):244-250.
- Rivalan, P., and coauthors. 2005. Trade-off between current reproductive effort and delay to next reproduction in the leatherback sea turtle. *Oecologia* 145(4):564-574.
- Rivas-Zinno, F. 2012. Captura incidental de tortugas marinas en Bajos del Solis, Uruguay. Universidad de la Republica Uruguay, Departamento de Ecologia y Evolucion.
- Rodrigues, J., D. Freitas, Í. Fernandes, and R. Lessa. 2015. Estrutura populacional do tubarão estrangeiro (*Carcharhinus longimanus*) no Atlântico Sul. 3.
- Rogers, S. G., and W. Weber. 1995. Status and restoration of Atlantic and shortnose sturgeons in Georgia, Final Report. National Marine Fisheries Service, Southeast Regional Office, St. Petersburg, Florida.

- Rosel, P. E., and coauthors. 2016. Status Review of Bryde's Whales (*Balaenoptera Edeni*) in the Gulf of Mexico under the Endangered Species Act. NOAA Technical Memorandum NMFS-SEFSC-692.
- Rosenthal, H., and D. F. Alderdice. 1976. Sub-lethal effects of environmental stressors, natural and pollutional, on marine fish eggs and larvae. *Journal of the Fisheries Research Board of Canada* 33:2047-2065.
- Ruben, H. J., and S. J. Morreale. 1999. Draft Biological Assessment for Sea Turtles in New York and New Jersey Harbor Complex. Unpublished Biological Assessment submitted to the National Marine Fisheries Service.
- Ruelle, R., and C. Henry. 1992. Organochlorine compounds in pallid sturgeon. *Contaminant Information Bulletin*.
- Ruelle, R., and K. D. Keenlyne. 1993. Contaminants in Missouri River pallid sturgeon. *Bulletin of Environmental Contamination and Toxicology* 50(6):898-906.
- Saba, V. S., and coauthors. 2016. Enhanced warming of the Northwest Atlantic Ocean under climate change. *Journal of Geophysical Research: Oceans* 121(1):118-132.
- SAFMC. 1998. Final Plan for the South Atlantic Region: Essential Fish Habitat Requirements for the Fishery Management Plan of the South Atlantic Fishery Management Council. South Atlantic Fishery Management Council, Charleston, SC.
- Sakai, H., H. Ichihashi, H. Suganuma, and R. Tatsukawa. 1995. Heavy metal monitoring in sea turtles using eggs. *Marine Pollution Bulletin* 30:347-353.
- Salwasser, H., S. P. Mealey, and K. Johnson. 1984. Wildlife population viability: a question of risk. Pages 421-439 in *Transactions of the North American Wildlife and Natural Resources Conference*.
- Sanches, J. G. 1991. Catálogo dos principais peixes marinhos da República de Guiné-Bissau. Publicações avulsas do I.N.I.P. No. 16. 429 p as cited in Froese, R. and D. Pauly, Editors. 2000. *FishBase 2000: concepts, design and data sources*. ICLARM, Los Baños, Laguna, Philippines. 344 p.
- Santidrián Tomillo, P., and coauthors. 2007. Reassessment of the leatherback turtle (*Dermochelys coriacea*) nesting population at Parque Nacional Marino Las Baulas, Costa Rica: Effects of conservation efforts. *Chelonian Conservation and Biology* 6(1):54-62.
- Sarti Martínez, L., and coauthors. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. *Chelonian Conservation and Biology* 6(1):70-78.
- Savoy, T. 2007. Prey eaten by Atlantic sturgeon in Connecticut waters. *American Fisheries Society Symposium* 56:157.

- Savoy, T., and D. Pacileo. 2003. Movements and important habitats of subadult Atlantic sturgeon in Connecticut waters. *Transactions of the American Fisheries Society* 132:1-8.
- Schmid, J. R., and J. A. Barichivich. 2006. *Lepidochelys kempii*—Kemp's ridley. Pages 128-141 in P. A. Meylan, editor. *Biology and conservation of Florida turtles*. Chelonian Research Monographs, volume 3.
- Schmid, J. R., and A. Woodhead. 2000. Von Bertalanffy growth models for wild Kemp's ridley turtles: analysis of the NMFS Miami Laboratory tagging database. U. S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, Florida.
- Scholz, N. L., and coauthors. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 57(9):1911-1918.
- Schroeder, B. A., and A. M. Foley. 1995. Population studies of marine turtles in Florida Bay. J. I. Richardson, and T. H. Richardson, editors. *Twelfth Annual Workshop on Sea Turtle Biology and Conservation*.
- Schueller, P., and D. L. Peterson. 2010. Abundance and recruitment of juvenile Atlantic sturgeon in the Altamaha River, Georgia. *Transactions of the American Fisheries Society* 139(5):1526-1535.
- Schultz, J. P. 1975. Sea turtles nesting in Surinam. *Zool. Verhand. Leiden* (143):172.
- Schulz, J. P. 1975. Sea turtles nesting in Surinam. *Zoologische Verhandelingen* 143:3-172.
- Schulz, J. P. 1982. Status of sea turtle populations nesting in Surinam with notes on sea turtles nesting in Guyana and French Guiana. Pages 435-438 in K. A. Bjorndal, editor. *Biology and Conservation of Sea Turtles*. Smithsonian Institution Press, Washington, D. C.
- Schulze-Haugen, M., T. Corey, and N. E. Kohler. 2003. *Guide to sharks, tunas, and billfishes of the U.S. Atlantic and Gulf of Mexico*. Rhode Island Sea Grant, University of Rhode Island.
- Scott, W. B., and E. J. Crossman. 1973a. Freshwater fishes of Canada. *Bulletin of the Fisheries Research Board of Canada* 184:1-966.
- Scott, W. B., and E. J. Crossman. 1973b. Freshwater fishes of Canada., *Fisheries Research Board of Canada Bulletin*.
- Secor, D. 1995. Chesapeake Bay Atlantic sturgeon: current status and future recovery. Summary of findings and recommendations from a workshop convened 8 November 1994 at Chesapeake Biological Laboratory. Chesapeake Bay Biological

Laboratory, Center for Estuarine and Environmental Studies, University of Maryland System, Solomons, MD.

- Secor, D. H. 2002. Atlantic sturgeon fisheries and stock abundances during the late nineteenth century. Pages 89-98 *in* American Fisheries Society Symposium.
- Secor, D. H., and T. E. Gunderson. 1998. Effects of hypoxia and temperature on survival, growth, and respiration of juvenile Atlantic sturgeon (*Acipenser oxyrinchus*). Fishery Bulletin U.S. 96:603-613.
- Secor, D. H., and coauthors. 2000. Dispersal and growth of yearling Atlantic sturgeon, *Acipenser oxyrinchus*, released into Chesapeake Bay. Fishery Bulletin 98(4):800-810.
- Secor, D. H., and J. R. Waldman. 1999. Historical Abundance of Delaware Bay Atlantic Sturgeon and Potential Rate of Recovery. Pages 203-216 *in* American Fisheries Society Symposium.
- SEFSC, N. a. 2011. Preliminary summer 2010 regional abundance estimate of loggerhead turtles (*Caretta caretta*) in northwestern Atlantic Ocean continental shelf waters. NEFSC Reference Document 11-03. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Reference Document 11-03, Woods Hole, Massachusetts.
- Seitz, J. C., and G. R. Poulakis. 2002. Recent Occurrence of Sawfishes (*Elasmobranchiomorphi: Pristidae*) Along the Southwest Coast of Florida (USA). Florida Scientist 65(4):11.
- Seitz, J. C., and G. R. Poulakis. 2006. Anthropogenic effects on the smalltooth sawfish (*Pristis pectinata*) in the United States. Marine Pollution Bulletin 52(11):1533-1540.
- Seki, T., T. Taniuchi, H. Nakano, and M. Shimizu. 1998. Age, growth and reproduction of the oceanic whitetip Shark from the Pacific Ocean. Fisheries Science 64:14-20.
- Seminoff, J. A., and coauthors. 2015. Status review of the green turtle (*Chelonia Mydas*) under the endangered species act. NOAA Technical Memorandum, NMFS-SWFSC-539.
- Shaffer, M. L. 1981. Minimum Population Sizes for Species Conservation. BioScience 31(2):131-134.
- Shaver, D. J. 1994. Relative abundance, temporal patterns, and growth of sea turtles at the Mansfield Channel, Texas. Journal of Herpetology 28(4):491-497.

- Shaver, D. J. 2002. Green sea turtles (*Chelonia mydas*) in a south Texas (USA) developmental habitat. Pages 9 in A. Mosier, A. Foley, and B. Brost, editors. Twentieth Annual Symposium on Sea Turtle Biology and Conservation.
- Shenker, J. M. 1984. Scyphomedusae in surface waters near the Oregon coast, May-August, 1981. *Estuarine, Coastal and Shelf Science* 19(6):619-632.
- Shepherd, G., and coauthors. 2007. Estimation of Atlantic Sturgeon bycatch in coastal Atlantic commercial fisheries of New England and the mid-Atlantic. Atlantic States Marine Fisheries Commission, Special Report to the Atlantic Sturgeon Management Board, Woods Hole, Massachusetts.
- Shillinger, G. L., and coauthors. 2008. Persistent leatherback turtle migrations present opportunities for conservation. *PLoS Biology* 6(7):1408-1416.
- Shoop, C. R., and R. D. Kenney. 1992. Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. *Herpetological Monographs* 6:43-67.
- Sidman, C. F., and coauthors. 2007. A Recreational Boating Characterization for Brevard County.
- Sidman, C. F., R. Swett, T. J. Fik, D. Fann, and W. B. Sargent. 2005. A Recreational Boating Characterization for the Greater Charlotte Harbor. Sea Grant.
- Silva Lee, A. F. 1974. Hábitos alimentarios de la cherna criolla *Epinephelus striatus* Bloch y algunos datos sobre su biología. *Serie Oceanologica Academia de Ciencias de Cuba* 25:3-14.
- Simpfendorfer, C. A. 2000. Predicting population recovery rates for endangered western Atlantic sawfishes using demographic analysis. *Environmental Biology of Fishes* 58(4):371-377.
- Simpfendorfer, C. A. 2001. Essential habitat of the smalltooth sawfish (*Pristis pectinata*). Report to the National Fisheries Service's Protected Resources Division. Mote Marine Laboratory Technical Report.
- Simpfendorfer, C. A. 2002. Smalltooth sawfish: The USA's first endangered *elasmobranch*. *Endangered Species Update* (19):53-57.
- Simpfendorfer, C. A. 2003. Abundance, movement and habitat use of the smalltooth sawfish. Final Report. Mote Marine Laboratory Mote Technical Report No. 929, Sarasota, FL.
- Simpfendorfer, C. A. 2006. Movement and habitat use of smalltooth sawfish. Final Report. Mote Marine Laboratory, Mote Marine Laboratory Technical Report 1070, Sarasota, FL.

- Simpfendorfer, C. A., G. R. Poulakis, P. M. O'Donnell, and T. R. Wiley. 2008. Growth rates of juvenile smalltooth sawfish, *Pristis pectinata* (Latham), in the western Atlantic. *Journal of Fish Biology* 72(3):711-723.
- Simpfendorfer, C. A., and T. R. Wiley. 2004a. Determination of the distribution of Florida's remnant sawfish population, and identification of areas critical to their conservation. Mote Marine Laboratory, Sarasota, Florida.
- Simpfendorfer, C. A., and T. R. Wiley. 2004b. Determination of the distribution of Florida's remnant sawfish population, and identification of areas critical to their conservation. Mote Marine Laboratory Technical Report. Mote Marine Laboratory, Sarasota, FL.
- Simpfendorfer, C. A., and T. R. Wiley. 2005a. Determination of the distribution of Florida's remnant sawfish population and identification of areas critical to their conservation. Final Report. Florida Fish and Wildlife Conservation Commission, Tallahassee, FL.
- Simpfendorfer, C. A., and T. R. Wiley. 2005b. Identification of priority areas for smalltooth sawfish conservation. Final report to the National Fish and Wildlife Foundation for Grant # 2003-0041-000. Mote Marine Laboratory.
- Simpfendorfer, C. A., T. R. Wiley, and B. G. Yeiser. 2010. Improving conservation planning for an endangered sawfish using data from acoustic telemetry. *Biological Conservation* 143:1460-1469.
- Simpfendorfer, C. A., and coauthors. 2011. Environmental Influences on the Spatial Ecology of Juvenile Smalltooth Sawfish (*Pristis pectinata*): Results from Acoustic Monitoring. *PLoS ONE* 6(2):e16918.
- Sindermann, C. J. 1994. Quantitative effects of pollution on marine and anadromous fish populations. NOAA Technical Memorandum NMFS-F/NEC-104. National Marine Fisheries Service, Woods Hole, Massachusetts.
- Skomal, G. B., B. C. Chase, and E. D. Prince. 2002. A comparison of circle hook and straight hook performance in recreational fisheries for juvenile Atlantic bluefin tuna. *Am. Fish. Soc. Symp.* 30:57-65.
- Smith, C. L. 1971. A revision of the American groupers: *Epinephelus* and allied genera. *Bulletin of the AMNH*; v. 146, article 2.
- Smith, J. A., H. J. Flowers, and J. E. Hightower. 2015. Fall spawning of Atlantic Sturgeon in the Roanoke River, North Carolina. *Transactions of the American Fisheries Society* 144(1):48-54.
- Smith, T. I. J. 1985. The fishery, biology, and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. *Environmental Biology of Fishes* 14(1):61-72.

- Smith, T. I. J., and J. P. Clugston. 1997. Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. *Environmental Biology of Fishes* 48(1-4):335-346.
- Smith, T. I. J., E. K. Dingley, and E. E. Marchette. 1980. Induced spawning and culture of Atlantic sturgeon *Progressive Fish Culturist* 42:147-151.
- Smith, T. I. J., D. E. Marchette, and R. A. Smiley. 1982. Life history, ecology, culture and management of Atlantic sturgeon, *Acipenser oxyrinchus*, Mitchell, in South Carolina. Final Report to U.S. Fish and Wildlife Service Resources Department.
- Smolowitz, R., H. Haas, H. O. Milliken, M. Weeks, and E. Matzen. 2010. Using sea turtle carcasses to assess the conservation potential of a turtle excluder dredge. *North American Journal of Fisheries Management* 30:993-1000.
- Snelson, F. F., and S. E. Williams. 1981. Notes on the occurrence, distribution, and biology of elasmobranch fishes in the Indian River Lagoon System, Florida. *Estuaries* 4(2):110-120.
- Snover, M. L. 2002. Growth and ontogeny of sea turtles using skeletochronology: Methods, validation and application to conservation. Duke University.
- Sobin, J. M. 2008. Diving Behavior of Female Loggerhead Turtles (*Caretta caretta*) During Their Internesting Interval and an Evaluation of the Risk of Boat Strikes. Masters project submitted in partial fulfillment of the requirements for the Master of Environmental Management degree in the Nicholas School of the Environment of Duke University. Dr. Scott Eckert, Advisor. December.
- Soulé, M. E. 1980. Thresholds for survival: maintaining fitness and evolutionary potential. Pages 151-170 in M. E. Soulé, and B. A. Wilcox, editors. *Conservation Biology: An Evolutionary-Ecological Perspective*. Sinauer Associates, Sunderland, MA.
- Soulé, M. E. 1987. Where do we go from here? Chapter 10 In: Soulé, M.E. (ed), *Viable Populations for Conservation*. Cambridge University Press, pp.175-183.
- Southwood, A. L., R. D. Andrews, F. V. Paladino, and D. R. Jones. 2005. Effects of diving and swimming behavior on body temperatures of Pacific leatherback turtles in tropical seas. *Physiological and Biochemical Zoology* 78:285-297.
- Spear, B. J. 2007. U.S. Management of Atlantic Sturgeon. *American Fisheries Society Symposium* 56:339-346.
- Spotila, J. 2004. *Sea Turtles: A Complete Guide to their Biology, Behavior, and Conservation*. Johns Hopkins University Press, Baltimore, Maryland.
- Spotila, J. R., and coauthors. 1996. Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? *Chelonian Conservation and Biology* 2(2):209-222.

- Spotila, J. R., R. D. Reina, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. 2000. Pacific leatherback turtles face extinction. *Nature* 405:529-530.
- Squiers, T. 2004. State of Maine 2004 Atlantic sturgeon compliance report to the Atlantic States Marine Fisheries Commission. Report submitted to Atlantic States Marine Fisheries Commission, December 22, 2004, Washington, D.C.
- Stabenau, E. K., and K. R. N. Vietti. 2003. The physiological effects of multiple forced submergences in loggerhead sea turtles (*Caretta caretta*). *Fishery Bulletin* 101(4):889-899.
- Stacy, B. A., J. L. Keene, and B. A. Schroder. 2016. Report of the Technical Expert Workshop: Developing National Criteria for Assessing Post-Interaction Mortality of Sea Turtles in Trawl, Net, and Pot/Trap Fisheries. U.S. Dept. of Commerce, NOAA, NOAA Technical Memorandum NMFS-OPR-53.
- Stapleton, S., and C. Stapleton. 2006. Tagging and nesting research on hawksbill turtles (*Eretmochelys imbricata*) at Jumby Bay, Long Island, Antigua, West Indies: 2005 annual report. Jumby Bay Island Company, Ltd.
- Starbird, C. H., A. Baldrige, and J. T. Harvey. 1993. Seasonal occurrence of leatherback sea turtles (*Dermochelys coriacea*) in the Monterey Bay region, with notes on other sea turtles, 1986-1991. *California Fish and Game* 79(2):54-62.
- Starbird, C. H., and M. M. Suarez. 1994. Leatherback sea turtle nesting on the north Vogelkop coast of Irian Jaya and the discovery of a leatherback sea turtle fishery on Kei Kecil Island. Pages 143-146 in K. A. Bjorndal, A. B. Bolten, D. A. Johnson, and P. J. Eliazar, editors. Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.
- Starr, R. M., E. Sala, E. Ballesteros, and M. Zabala. 2007. Spatial dynamics of the Nassau grouper *Epinephelus striatus* in a Caribbean atoll. *Marine Ecology Progress Series* 343:239-249.
- Stedman, S., and T. E. Dahl. 2008. Status and trends of wetlands in the coastal watersheds of the Eastern United States 1998-2004. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, and U.S. Department of the Interior, U.S. Fish and Wildlife Service.
- Stein, A. B., K. D. Friedland, and M. Sutherland. 2004a. Atlantic sturgeon marine bycatch and mortality on the continental shelf of the Northeast United States. *North American Journal of Fisheries Management* 24(1):171-183.
- Stein, A. B., K. D. Friedland, and M. Sutherland. 2004b. Atlantic sturgeon marine distribution and habitat use along the northeastern coast of the United States. *Transactions of the American Fisheries Society* 133:527-537.

- Stein, A. B., K. D. Friedland, and M. Sutherland. 2004c. Atlantic Sturgeon Marine Distribution and Habitat Use along the Northeastern Coast of the United States. *Transactions of the American Fisheries Society* 133(3):527-537.
- Stevens, J. D., and J. M. Lyle. 1989. Biology of 3 hammerhead sharks (*Eusphyra blochii*, *Sphyrna mokarran* and *S. lewini*) from northern Australia. *Australian Journal of Marine and Freshwater Research* 40(2):129-146.
- Stevenson, J. C., and D. H. Secor. 1999. Age determination and growth of Hudson River Atlantic sturgeon (*Acipenser oxyrinchus*). *Fishery Bulletin* 97:153-166.
- Stewart, K., and C. Johnson. 2006. *Dermochelys coriacea*—Leatherback sea turtle. *Chelonian Research Monographs* 3:144-157.
- Stewart, K., C. Johnson, and M. H. Godfrey. 2007. The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. *Herpetological Journal* 17(2):123-128.
- Steyermark, A. C., and coauthors. 1996. Nesting leatherback turtles at Las Baulas National Park, Costa Rica. *Chelonian Conservation and Biology* 2(2):173-183.
- Stokes, L., and coauthors. 2006. Evaluation of injury potential in incidentally captured loggerhead sea turtles (*Caretta caretta*) relating to hook size and baiting technique. Pages 267 in Frick, M. A. Panagopoulou, A. F. Rees, and K. Williams (compilers), editors. *Book of Abstracts, 26th Annual Symposium on Sea Turtle Biology and Conservation*, Island of Crete, Greece April 3-8.
- Storelli, M. M., G. Barone, A. Storelli, and G. O. Marcotrigiano. 2008a. Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. *Chemosphere* 70(5):908-913.
- Storelli, M. M., G. Barone, A. Storelli, and G. O. Marcotrigiano. 2008b. Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. *Chemosphere* 70:908-913.
- Storelli, M. M., E. Ceci, and G. O. Marcotrigiano. 1998. Distribution of heavy metal residues in some tissues of *Caretta caretta* (Linnaeus) specimen beached along the Adriatic Sea (Italy). *Bulletin of Environmental Contamination and Toxicology* 60:546-552.
- Strasburg, D. W. 1958. Distribution, abundance, and habits of pelagic sharks in the central Pacific Ocean. *Fisheries* 1:2S.
- Suchman, C., and R. Brodeur. 2005. Abundance and distribution of large medusae in surface waters of the northern California Current. *Deep Sea Research Part II: Topical Studies in Oceanography* 52(1-2):51-72.

- Sweka, J., and coauthors. 2007. Juvenile Atlantic sturgeon habitat use in Newburgh and Haverstraw Bays of the Hudson River: Implications for Population Monitoring. *North American Journal of Fisheries Management* 27:1058-1067.
- Tambourgi, M., and coauthors. 2013. Reproductive aspects of the oceanic whitetip shark, *Carcharhinus longimanus* (Elasmobranchii: Carcharhinidae), in the equatorial and southwestern Atlantic Ocean. *Brazilian Journal of Oceanography* 61:161-168.
- TEWG. 1998a. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the Western North Atlantic. Department of Commerce, Turtle Expert Working Group.
- TEWG. 1998b. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the western North Atlantic. U. S. Dept. Commerce.
- TEWG. 2000a. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Turtle Expert Working Group.
- TEWG. 2000b. Assessment update for the kemp's ridley and loggerhead sea turtle populations in the western North Atlantic : a report of the Turtle Expert Working Group. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, Fla.
- TEWG. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Turtle Expert Working Group.
- TEWG. 2009. An assessment of the loggerhead turtle population in the western North Atlantic ocean. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Turtle Expert Working Group, NMFS-SEFSC-575.
- Thomas, C. D. 1990. What Do Real Population Dynamics Tell Us About Minimum Viable Population Sizes? *Conservation Biology* 4(3):324-327.
- Thompson, R., and J. L. Munro. 1978. Aspects of the biology and ecology of Caribbean reef fishes: Serranidae (hinds and groupers). *Journal of Fish Biology* 12:115-146.
- Thorpe, T., and D. Bereshoff. 2000. Determination of gillnet bycatch potential of spiny dogfish (*Squalus acanthias* L.) in southeastern North Carolina. Completion Report, North Carolina State University, North Carolina Sea Grant, Fisheries Resource Grant Program, Grant number 99-FEG-47, Raleigh.
- Thorson, T. B. 1976. Observations on the reproduction of the sawfish *Pristis perotteti*, in Lake Nicaragua, with recommendations for its conservation. T. B. Thorson, editor.

Investigations of the Ichthyofauna of Nicaraguan Lakes. Univ. Nebraska, Lincoln, NB.

- Thorson, T. B. 1982. Life history implications of a tagging study of the largemouth sawfish, *Pristis perotteti*, in the Lake Nicaragua-Río San Juan system. *Environmental Biology of Fishes* 7(3):207-228.
- Tolotti, M. T., P. Bach, F. Hazin, P. Travassos, and L. Dagorn. 2015. Vulnerability of the Oceanic Whitetip Shark to Pelagic Longline Fisheries. *PLoS ONE* 10(10).
- Trencia, G., G. Verreault, S. Georges, and P. Pettigrew. 2002. Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) fishery management in Québec, Canada, between 1994 and 2000. *Journal of Applied Ichthyology* 18(4-6):455-462.
- Troëng, S. 1998. Poaching threatens the green turtle rookery at Tortuguero, Costa Rica. *Marine Turtle Newsletter* 80(11-12).
- Troeng, S., D. Chacon, and B. Dick. 2004. Leatherback turtle *Dermochelys coriacea* nesting along the Caribbean coast of Costa Rica. Pages 13 in M. S. Coyne, and R. D. Clark, editors. *Twenty-First Annual Symposium on Sea Turtle Biology and Conservation*.
- Troëng, S., D. Chacón, and B. Dick. 2004. Possible decline in leatherback turtle *Dermochelys coriacea* nesting along the coast of Caribbean Central America. *Oryx* 38:395-403.
- Troeng, S., D. R. Evans, E. Harrison, and C. J. Lagueux. 2005. Migration of green turtles *Chelonia mydas* from Tortuguero, Costa Rica. *Marine Biology* 148(2):435-447.
- Troëng, S., E. Harrison, D. Evans, A. d. Haro, and E. Vargas. 2007. Leatherback turtle nesting trends and threats at Tortuguero, Costa Rica. *Chelonian Conservation and Biology* 6(1):117-122.
- Troëng, S., and E. Rankin. 2005. Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. *Biological Conservation* 121:111-116.
- Tucker, A. D. 1988. A summary of leatherback turtle *Dermochelys coriacea* nesting at Culebra, Puerto Rico from 1984-1987 with management recommendations. U.S. Fish and Wildlife Service.
- Tucker, A. D. 2010. Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: Implications for stock estimation. *Journal of Experimental Marine Biology and Ecology* 383(1):48-55.

- Tucker, J. W., P. G. Bush, and S. T. Slaybaugh. 1993. Reproductive patterns of Cayman Islands Nassau grouper (*Epinephelus striatus*) populations. *Bulletin of Marine Science* 52(3):961-969.
- USFWS. 2003. Fish and Wildlife Coordination Act Report on Savannah River Basin Comprehensive Study. United States Fish and Wildlife Service.
- USFWS and NMFS. 1998. Endangered Species Consultation Handbook. Procedures for Conducting Section 7 Consultations and Conferences. U.S. Fish and Wildlife Service and National Marine Fisheries Service, March 1998.
- USGRG. 2004. U.S. National Assessment of the Potential Consequences of Climate Variability and Change, Regional Paper: The Southeast. U.S. Global Research Group. Washington, D.C., August 20, 2004.
- van Dam, R., and L. Sarti. 1989. Sea Turtle Biology and Conservation on Mona Island, Puerto Rico. Report for 1989. Chelonia, Sociedad Herpetologica de Puerto Rico, San Juan, Puerto Rico.
- Van Dam, R., L. Sarti M., and D. Pares J. 1991. The hawksbills of Mona Island, Puerto Rico: Report for 1990. Sociedad Chelonia and Departamento. Recursos Naturales, Puerto Rico.
- Van Dam, R. P., and C. E. Diez. 1997. Predation by hawksbill turtles on sponges at Mona Island, Puerto Rico. Pages 1421-1426 *in* Eighth International Coral Reef Symposium.
- Van Dam, R. P., and C. E. Diez. 1998. Home range of immature hawksbill turtles (*Eretmochelys imbricata* (Linnaeus)) at two Caribbean islands. *Journal of Experimental Marine Biology and Ecology* 220:15-24.
- Van Eenennaam, J. P., and S. I. Doroshov. 1998. Effects of age and body size on gonadal development of Atlantic sturgeon. *Journal of Fish Biology* 53(3):624-637.
- Van Eenennaam, J. P., and coauthors. 1996. Reproductive Conditions of the Atlantic Sturgeon (*Acipenser oxyrinchus*) in the Hudson River. *Estuaries* 19(4):769-777.
- Vannuccini, S. 1999. Shark utilization, marketing, and trade. Food & Agriculture Org.
- Vermeer, M., and S. Rahmstorf. 2009. Global sea level linked to global temperature. *Proceedings of the National Academy of Sciences* 106(51):21527-21532.
- Vladykov, V. D., and J. R. Greely. 1963a. Order Acipenseroidei. *Fishes of Western North Atlantic*. Yale.
- Vladykov, V. D., and J. R. Greely. 1963b. Order Acipenseroidei. Pages 1630 pp *in* *Fishes of Western North Atlantic*, Sears Foundation. Marine Research, Yale University.

- Von Westernhagen, H., and coauthors. 1981a. Bioaccumulating substances and reproductive success in baltic flounder (*Platichthys flesus*). *Aquatic Toxicology* 1(2):85-99.
- Von Westernhagen, H., and coauthors. 1981b. Bioaccumulating substances and reproductive success in baltic flounder platichthys flesus. *Aquatic Toxicology* 1(2):85-99.
- Waldman, J. R., C. Grunwald, J. Stabile, and I. Wirgin. 2002. Impacts of life history and biogeography on the genetic stock structure of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*, Gulf sturgeon *A. oxyrinchus desotoi*, and shortnose sturgeon *A. brevirostrum*. *Journal of Applied Ichthyology* 18(4-6):509-518.
- Waldman, J. R., and I. I. Wirgin. 1998. Status and restoration options for Atlantic sturgeon in North America. *Conservation Biology* 12(3):631-638.
- Wallace, B. P., J. Schumacher, J. A. Seminoff, and M. C. James. 2014. Biological and environmental influences on the trophic ecology of leatherback turtles in the northwest Atlantic Ocean. *Marine Biology*.
- Warden, M. L., and K. T. Murray. 2011. Reframing protected species interactions with commercial fishing gear: Moving toward estimating the unobservable. *Fisheries Research* 110(3):387-390.
- Waring, C. C., L. M. Moller, and R. G. Harcourt. 2013. Size stability of the heavily touristed population of Indo-Pacific bottlenose dolphins of Port Stephens, eastern Australia. Pages 220-221 in *Twentieth Biennial Conference on the Biology of Marine Mammals*, Dunedin, New Zealand.
- Waring, C. P., and A. Moore. 2004. The effect of atrazine on Atlantic salmon (*Salmo salar*) smolts in fresh water and after sea water transfer. *Aquatic Toxicology* 66(1):93-104.
- Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel. 2012. US Atlantic and Gulf of Mexico marine mammal stock assessments - 2011. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.
- Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster. 2005. Fishing methods to reduce sea turtle mortality associated with pelagic longlines. *Canadian Journal of Fisheries and Aquatic Sciences* 62(5):965-981.
- Weber, W., and C. A. Jennings. 1996. Endangered species management plan for the shortnose sturgeon, *Acipenser brevirostrum*. Final Report to Port Stewart Military Reservation, Fort Stewart, GA.

- Weijerman, M. L., H. G. V. Tienen, A. D. Schouten, and W. E. J. Hoekert. 1998. Sea turtles of Galibi, Suriname. Pages 142-144 in R. Byles, and Y. Fernandez, editors. Sixteenth Annual Symposium on Sea Turtle Biology and Conservation.
- Weishampel, J. F., D. A. Bagley, and L. M. Ehrhart. 2004. Earlier nesting by loggerhead sea turtles following sea surface warming. *Global Change Biology* 10:1424-1427.
- Weishampel, J. F., D. A. Bagley, L. M. Ehrhart, and B. L. Rodenbeck. 2003. Spatiotemporal patterns of annual sea turtle nesting behaviors along an East Central Florida beach. *Biological Conservation* 110(2):295-303.
- Welsh, S. A., S. M. Eyler, M. F. Mangold, and A. J. Spells. 2002. Capture locations and growth rates of Atlantic sturgeon in the Chesapeake Bay. Pages 183-194 in American Fisheries Society Symposium.
- Wenzel, F. W., D. K. Mattila, and P. J. Clapham. 1988. *Balaenoptera musculus* in the Gulf of Maine. *Marine Mammal Science* 4(2):172-175.
- Wershoven, J. L., and R. W. Wershoven. 1992. Juvenile green turtles in their nearshore habitat of Broward County, Florida: A five year review. Pages 121-123 in M. Salmon, and J. Wyneken, editors. Eleventh Annual Workshop on Sea Turtle Biology and Conservation.
- White, F. N. 1994. Swallowing dynamics of sea turtles. Pages 89-95 in G. H. Balazs, and S. G. Pooley, editors. Research Plan to Assess Marine Turtle Hooking Mortality. National Oceanic and Atmospheric Administration, Honolulu, Hawaii.
- Whitfield, A. K., and M. N. Bruton. 1989. Some biological implications of reduced freshwater inflow into eastern Cape estuaries: a preliminary assessment. *South African Journal of Science* 85:691-694.
- Whiting, S. D. 2000. The foraging ecology of juvenile green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) sea turtles in north-western Australia. Northern Territory University, Darwin, Australia.
- Wiley, T. R., and C. A. Simpfendorfer. 2007. The ecology of elasmobranchs occurring in the Everglades National Park, Florida: implications for conservation and management. *Bulletin of Marine Science* 80(1):171-189.
- Wilkinson, C. 2004. Status of Coral Reefs of the World: 2004. Australian Institute of Marine Science, ISSN 1447-6185.
- Winger, P. V., P. J. Lasier, D. H. White, and J. T. Seginak. 2000. Effects of Contaminants in Dredge Material from the Lower Savannah River. *Archives of Environmental Contamination and Toxicology* 38(1):128-136.

- Wirgin, I., C. Grunwald, J. Stabile, and J. Waldman. 2007. Genetic evidence for relict Atlantic sturgeon stocks along the mid-Atlantic coast of the USA. *North American Journal of Fisheries Management* 27(4):1214-1229.
- Wirgin, I., and T. King. 2011. Mixed stock analysis of Atlantic sturgeon from coastal locales and a non-spawning river. NMFS Northeast Region Sturgeon Workshop, Alexandria, Virginia.
- Wirgin, I., L. Maceda, C. Grunwald, and T. L. King. 2015. Population origin of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* by-catch in US Atlantic coast fisheries. *Journal of Fish Biology* 86(4):1251-1270.
- Wirgin, I., J. Waldman, J. Stabile, B. Lubinski, and T. King. 2002. Comparison of mitochondrial DNA control region sequence and microsatellite DNA analyses in estimating population structure and gene flow rates in Atlantic sturgeon *Acipenser oxyrinchus*. *Journal of Applied Ichthyology* 18(4-6):313-319.
- Wirgin, I., and coauthors. 2000. Genetic structure of Atlantic sturgeon populations based on mitochondrial DNA control region sequences. *Transactions of the American Fisheries Society* 129(2):476-486.
- Witherington, B., M. Bresette, and R. Herren. 2006. *Chelonia mydas* - Green turtle. *Chelonian Research Monographs* 3:90-104.
- Witherington, B., S. Hirama, and A. Moiser. 2003. Effects of beach armoring structures on marine turtle nesting. U.S. Fish and Wildlife Service.
- Witherington, B., S. Hirama, and A. Moiser. 2007. Changes to armoring and other barriers to sea turtle nesting following severe hurricanes striking Florida beaches. U.S. Fish and Wildlife Service.
- Witherington, B. E. 1992. Behavioral responses of nesting sea turtles to artificial lighting. *Herpetologica* 48(1):31-39.
- Witherington, B. E. 2002. Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. *Marine Biology* 140(4):843-853.
- Witherington, B. E., and K. A. Bjorndal. 1991. Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. *Biological Conservation* 55(2):139-149.
- Witherington, B. E., and L. M. Ehrhart. 1989a. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon System, Florida. *Copeia* 1989(3):696-703.
- Witherington, B. E., and L. M. Ehrhart. 1989b. Status, and reproductive characteristics of green turtles (*Chelonia mydas*) nesting in Florida. Pages 351-352 in L. Ogren, and coeditors, editors. Second Western Atlantic Turtle Symposium. .

- Witt, M. J., and coauthors. 2007. Prey landscapes help identify foraging habitats for leatherback turtles in the NE Atlantic. *Marine Ecology Progress Series* 337:231-243.
- Witt, M. J., B. J. Godley, A. C. Broderick, R. Penrose, and C. S. Martin. 2006. Leatherback turtles, jellyfish and climate change in the northwest Atlantic: Current situation and possible future scenarios. Pages 356-357 in M. Frick, A. Panagopoulou, A. F. Rees, and K. Williams, editors. *Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation*. International Sea Turtle Society, Athens, Greece.
- Witzell, W. N. 1983. Synopsis of biological data on the hawksbill sea turtle, *Eretmochelys imbricata* (Linnaeus, 1766). Food and Agricultural Organization of the United Nations, Rome.
- Witzell, W. N. 2002. Immature Atlantic loggerhead turtles (*Caretta caretta*): Suggested changes to the life history model. *Herpetological Review* 33(4):266-269.
- Work, T. M. 2000. Synopsis of necropsy findings of sea turtles caught by the Hawaii-based pelagic longline fishery.
- Wrona, A., and coauthors. 2007. Restoring ecological flows to the lower Savannah River: a collaborative scientific approach to adaptive management. Pages 27-29 in *Proceedings of the 2007 Georgia Water Resources Conference*.
- Young, C. N., and coauthors. 2016. Status review report: oceanic whitetip shark (*Carcharhinus longimanus*). Final Report to the National Marine Fisheries Service, Office of Protected Resources. .
- Young, J. R., T. B. Hoff, W. P. Dey, and J. G. Hoff. 1988. Management recommendations for a Hudson River Atlantic sturgeon fishery based on an age-structured population model. *Fisheries Research in the Hudson River*. State of University of New York Press, Albany, New York.
- Zug, G. R., and R. E. Glor. 1998. Estimates of age and growth in a population of green sea turtles (*Chelonia mydas*) from the Indian River lagoon system, Florida: A skeletochronological analysis. *Canadian Journal of Zoology* 76(8):1497-1506.
- Zug, G. R., and J. F. Parham. 1996. Age and growth in leatherback turtles, *Dermochelys coriacea*: A skeletochronological analysis. *Chelonian Conservation and Biology* 2:244-249.
- Zurita, J. C., and coauthors. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. Pages 25-127 in J. A. Seminoff, editor *Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation*, Miami, Florida.
- Zurita, J. C., B. Prezias, R. Herrera, and J. L. Miranda. 1994. Sea turtle tagging program in Quintana Roo, Mexico. Pages 300-303 in K. A. Bjorndal, A. B. Bolten, D. A.

Johnson, and P. J. Eliazar, editors. Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.

Zwinenberg, A. J. 1977. Kemp's ridley, *Lepidochelys kempii* (Garman, 1880), undoubtedly the most endangered marine turtle today (with notes on the current status of *Lepidochelys olivacea*). Bulletin Maryland Herpetological Society 13(3):170-192.

Appendix A. Anticipated Incidental Take of ESA-Listed Species in Federal Fisheries

Table A.1 Anticipated Incidental Takes of Sea Turtles in Federal Fisheries

Fishery	ITS Authorization Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
American Lobster [NER]	1 Year	1-Lethal or nonlethal	7Lethal or nonlethal	None	None	None
Batched Consultation* (gillnet) [NER]	1 Year	269-No more than 167 lethal (Takes based on a 5-yr average)	4-No more than 3 lethal	4-No more than 3 lethal	4-No more than 3 lethal	None
Batched Consultation* (bottom trawl) [NER]	1 Year	213-No more than 71 lethal (Takes based on a 4-yr average)	4-No more than 2 lethal	3-No more than 2 lethal	3-No more than 2 lethal	None
Batched Consultation* (trap/pot) [NER]	1 Year	1-Lethal or nonlethal	4-Lethal or nonlethal	None	None	None
Caribbean Reef Fish [SER]	3 Years	None	18-All lethal	None	75-All lethal	51-No more than 3 lethal
Coastal Migratory Pelagics [SER]	3 Years	27 Total, 7 lethal	1- Lethal	8- Total, 2 lethal	31-Total, 9 lethal	1- Lethal
Dolphin-Wahoo [SER]	1 Year	12-No more than 2 lethal	12-No more than 1 lethal	3 for all species in combination-no more than 1 lethal take		
Gulf of Mexico Reef Fish [SER]	3 Years	1,044-No more than 572 lethal	11-All lethal	108-No more than 41 lethal	116-No more than 75 lethal	9-No more than 8 lethal

* Batched consultation includes the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass Fisheries

Table A.1 Anticipated Incidental Takes of Sea Turtles in Federal Fisheries, continued

Fishery	ITS Authorization Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
HMS-Shark Fisheries [SER] (Until this Current Consultation)	3 Years	126-No more than 78 lethal	18-No more than 9 lethal	36-No more than 21 lethal	57-No more than 33 lethal	18-No more than 9 lethal
HMS-Other Fisheries [SER] (Until this Current Consultation)	3 Years	No more than 3 sea turtles, of any species, in combination, per calendar year.				
HMS-Pelagic Longline [SER]	3 Years	1,905-No more than 339 lethal	1,764-No more than 252 lethal	105-No more than 18 lethal for these species in combination		
Red Crab [NER]	1 Year	1-Lethal or nonlethal	1-Lethal or nonlethal	None	None	None
Caribbean Spiny Lobster	3 Years	None	9 – Lethal or non-lethal	None	12- Lethal or non-lethal	12 – Lethal or non-lethal take
Gulf of Mexico/South Atlantic Spiny Lobster Fishery [SER]	3 Years	3-Lethal or Nonlethal Take	1 –Lethal or Nonlethal take for Leatherbacks, Hawksbill, and Kemp's ridley		3-Lethal or Nonlethal Take	1 –Lethal or Nonlethal take for Leatherbacks, Hawksbill, and Kemp's ridley
South Atlantic Snapper-Grouper [SER]	3 Years	629-No more than 208 lethal	6-No more than 5 lethal	180-No more than 59 lethal	NA DPS – 111-No more than 42 lethal SA DPS - 6-No more than 3 lethal	6-No more than 4 lethal
Southeastern U.S. Shrimp [SER]	1 Year	Anticipated shrimp trawl effort (i.e., 132,900 days fished in the Gulf of Mexico and 14,560 trips in the south Atlantic) and fleet TED compliance (i.e., compliance resulting in overall average sea turtle catch rates in the shrimp otter trawl fleet at or below 12%) are used as surrogates for numerical sea turtle take levels.				

Table A.1 Anticipated Incidental Takes of Sea Turtles in Federal Fisheries, continued

Fishery	ITS Authorization Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
Atlantic Sea Scallop – Dredge [NER]	1 Year	161 – No more than 46 lethal	2 –Lethal Takes (gears combined)	3 – No more than 2 Lethal (gears combined)	2 - Lethal takes (gears combined)	None
Atlantic Sea Scallop – Trawl [NER]	1 Year	140 – No more than 66 lethal				None

* Batched consultation includes the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass Fisheries

Table A.2. Anticipated Incidental Take of Smalltooth Sawfish in Federal Fisheries

Fishery	3-Year Incidental Take of Smalltooth Sawfish
ATLANTIC HMS-SHARK FISHERIES (UNTIL THIS CURRENT CONSULTATION)	32– No more than 7 lethal takes
COASTAL MIGRATORY PELAGICS	1 Nonlethal takes
GULF OF MEXICO/SOUTH ATLANTIC SPINY LOBSTER FISHERY	2 Nonlethal takes
GULF OF MEXICO REEF FISH	8 Nonlethal takes
SOUTH ATLANTIC SNAPPER-GROUPER	8 Nonlethal takes
SOUTHEASTERN U.S. SHRIMP	288– No more than 105 lethal takes

Table A.3. Anticipated Incidental Take of Atlantic Sturgeon by DPS in Federal Fisheries

Fishery	ITS Authorizat ion Period	Atlantic Sturgeon DPS				
		Gulf of Maine	New York Bight	Chesapeake Bay	Carolina	South Atlantic
Southeastern U.S. Shrimp [SER]	3 years	Up to 162 interactions - including 27 captures, no more than 3 lethal	Up to 465 interactions – including 66 captures, no more than 9 lethal	Up to 312 interactions – including 54, no more than 6 lethal	Up to 519 interactions – including 87 captures, no more than 9 lethal	Up to 1,404 interactions – including 228 captures, no more than 21 lethal
HMS Shark and Smoothhound [SER] (Until this Current Consultation)	3 years	36-No more than 7 lethal	159-No more than 30 lethal	45-No more than 9 lethal	63-No more than 12 lethal	18-No more than 6 lethal
Coastal Migratory Pelagic Fishery [SER]	3 years	2, No lethal	4, No lethal	3, No lethal	4, No lethal	10, No lethal
Batched Consultation* (gillnet) [NER]	1 year (Takes based on a 5-yr average)	137-No more than 17 lethal A.E.s	632-No more than 79 lethal A.E.s	162-No more than 21 lethal A.E.s	25-No more than 4 lethal A.E.s	273-No more than 34 lethal A.E.s
Batched Consultation* (bottom trawl) [NER]	1 year (Takes based on a 5-yr average)	148-No more than 5 lethal A.E.s	685-No more than 21 lethal A.E.s	175-No more than 6 lethal A.E.s	27-No more than 1 lethal A.E.s	296-No more than 6 lethal A.E.s
Atlantic Sea Scallop Dredge [NER]	20 years	1 – Lethal (any DPS)				

A.E. = Adult equivalents

Appendix B Requirements for Handling Incidentally Taken Sturgeon and Collecting Genetic Samples

General Handling of Sturgeon

1. If the animal appears energetic, active, and otherwise healthy enough to undergo handling, it should be done so in accordance with guideline #3 below. If the animal is not healthy enough to undergo the procedures described, ensure the vessel is in neutral and release it over the side, head first.
2. Animals should be handled rapidly, but with care and kept in water to the maximum extent possible during holding and handling. During handling procedures the animal must be kept wet at all times using water from which it was removed (e.g., river water). While moving the animal or removing it from gear, covering its eyes with a wet towel may help calm it.
3. All handling procedures (i.e., measuring, PIT tagging, photographing, and tissue sampling) should be completed as quickly as possible, and should not exceed 20 minutes from when the sturgeon is first brought on board the vessel. Handling procedures should be prioritize in the following order: 1) collect a tissue sample (see procedure described below); 2) scan for existing PIT tags, apply new PIT tag if no pre-existing PIT tag is found; 3) measure the animal; 4) photograph the animal. If all of the handling procedures cannot be completed within 20 minutes, the animal should be returned to the water; indicate which procedures were not completed when reporting the incidental take to NMFS.
4. A sturgeon maybe held on board for longer than 20 minutes only when held in a net pen/basket floating next to the vessel or placed in flow through tanks, where the total volume of water is replaced every 15-20 minutes.

Genetic Tissue Sampling for Atlantic Sturgeon

5. Genetic tissue samples must be taken from every Atlantic sturgeon captured unless conditions are such that collecting a sample would imperil human or animal safety.
6. Tissue samples should be a small (1.0 cm²) fin clip collected from soft pelvic fin tissue. Use a knife, scalpel, or scissors that has been thoroughly cleaned and wiped with alcohol. Samples should be preserved in RNAlater™ preservative. Gently shake to ensure the solution covers the fin clip. Once the fin clip is in buffer solution, refrigeration/freezing is not required, but care should be taken not to expose the sample to excessive heat or intense sunlight. Label each sample with the fish's unique ID number. Do not use glass vials; a 2 ml screw top plastic vial is preferred (e.g., MidWest Scientific AVFS2002 and AVC100N).

PIT Tagging

7. Every sturgeon should be scanned for PIT tags along its entire body surface ensuring it has not been previously tagged. The PIT tag readers must be able to read both 125 kHz and 134 kHz tags. When a previously implanted tag is detected the PIT tag information should be recorded on the reporting spreadsheet ("Sturgeon Genetic Sample Submission

sheet”). Indicate the animal was a recapture in the “comment” field of the reporting spreadsheet. A copy of that reporting spreadsheet should be sent to mike_mangold@fws.gov.

8. Sturgeon without an existing PIT tag should have one implanted. The recommended frequency for PIT tags is 134.2 kHz. The tag information should be reported in the appropriate fields on the reporting spreadsheet.
9. Sturgeon smaller than 250mm shall not be PIT tagged. Sturgeon measuring 250-350 mm TL shall only be tagged with 8mm PIT tags. Sturgeon 350 mm or greater shall receive standard sized PIT tags (e.g., 11 or 14 mm).
10. PIT tags should be implanted to the left of the spine immediately anterior to the dorsal fin, and posterior to the dorsal scutes (Figure 1). This positioning optimizes the PIT tag’s readability over the animal’s lifetime. If necessary, to ensure tag retention and prevent harm or mortality to small juvenile sturgeon of all species, the PIT tag can also be inserted at the widest dorsal position just to the left of the 4th dorsal scute.
11. Scan the newly implanted tag following insertion to ensure it is readable before the animal is released. If the tag is not readable, one additional tag should be implanted on the opposite side following the same procedure, if doing so will not jeopardize the safety of the animal.

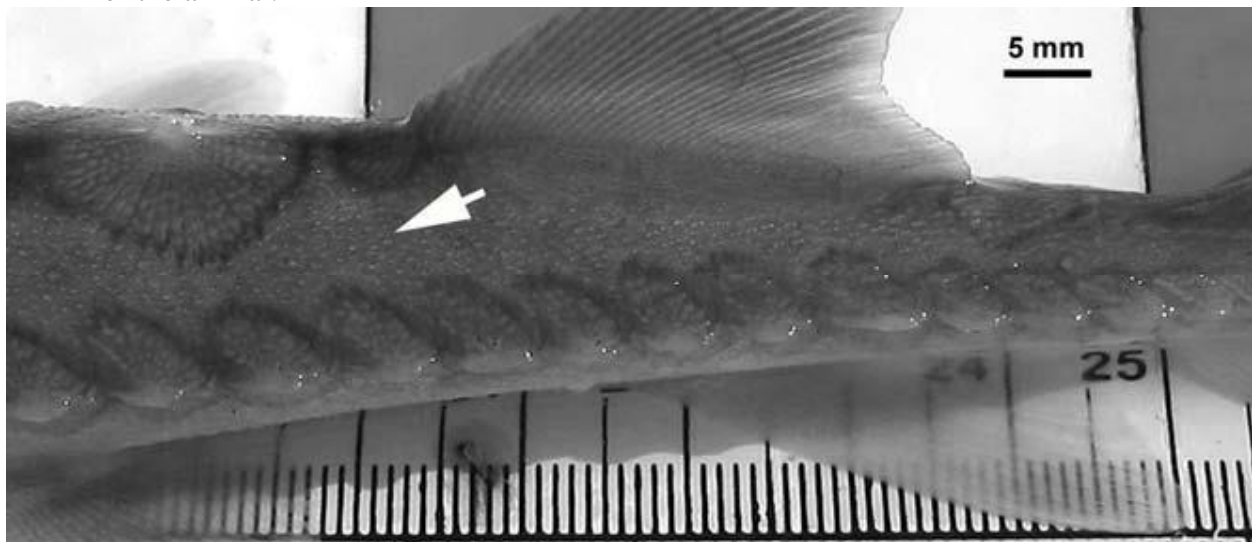


Figure 1. Standardized Location for PIT Tagging all Gulf, Atlantic, and shortnose sturgeon (Photo Credit: J. Henne, USFWS)

Measuring

12. Length measurements for all sturgeon should be taken as a straight line measurement from the snout to the fork in the tail (i.e., fork length – FL), and as a straight line measurement from the snout to the tip of the tail (i.e., total length – TL) (Figure 2). Do not measure the curve of the animal's body.

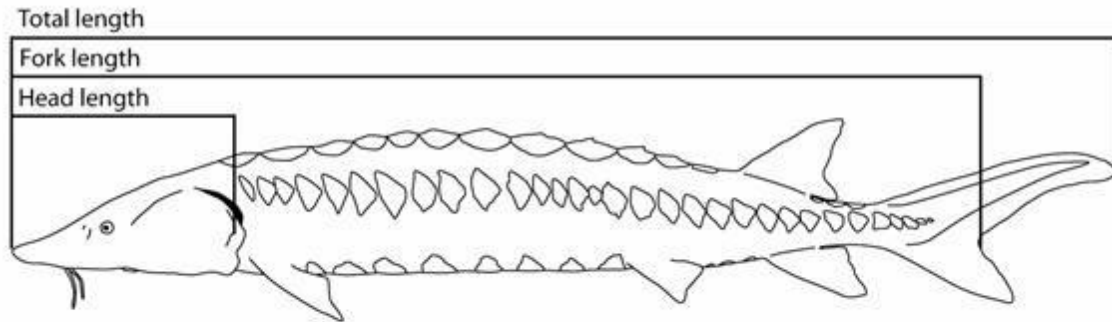


Figure 2. Diagram of different types of measurements for sturgeons.
(Drawings by Eric Hilton, Virginia Institute of Marine Science in Mohead and Kahn 2010)

Reporting Captures/Samples

13. *Reporting Captures and Genetic Samples*: Incidental captures and genetic samples may be reported using the same reporting spreadsheet (“Sturgeon Genetic Sample Submission Sheet”). Electronic metadata for each sample must be provided to properly identify and archive samples. Submit the reporting spreadsheet via email to: rjohnson1@usgs.gov and takereport.nmfs@noaa.gov. When submitting electronic metadata samples, identify the project name and biological opinion (SER #) in the subject line.
14. *Reporting Captures with NO Genetic Sample*: If no genetic sample could be safely collected, the incidental capture must still be reported using the Sturgeon Genetic Sample Submission Sheet. Submit the reporting spreadsheet via email to takereport.nmfs@noaa.gov. When submitting electronic metadata samples, identify the project name and biological opinion (SER #) in the subject line.

Transport of Genetic Samples

15. Package vials containing genetic samples together (e.g., in one box) with an absorbent material within a double-sealed container (e.g., zip lock baggie).
16. When submitting tissue samples via mail, identify the project name and biological opinion (SER #) under which the take was authorized in the shipping container. Ship tissue samples to: Robin Johnson, Geological Survey, Leetown Science Center, Aquatic Ecology Branch, 11649 Leetown Road, Kearneysville, WV 25430