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 Pelagic Longline Fishery for Atlantic Highly Migratory Species (F/SER/2014/00006[13697])

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

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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 the Pelagic Longline Fishery for Atlantic Highly Migratory Species (HMS), as managed under the 2006 Consolidated Atlantic HMS Fishery Management Plan (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 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, 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

NMFS has conducted numerous formal and informal ESA Section 7 consultations on Atlantic Highly Migratory fisheries, currently managed under the 2006 Consolidated Atlantic HMS FMP and associated amendments. The 2006 Consolidated HMS FMP merged under one plan the management of fisheries targeting Atlantic HMS, including Atlantic billfish, swordfish, tunas, and sharks. Previously, those fisheries were managed under the 1999 Atlantic HMS FMP (1999 HMS FMP) (swordfish, tunas, and sharks) and the 1999 Atlantic Billfish FMP (1999 Billfish FMP) (billfish). Consultations have analyzed all components of the Atlantic HMS fisheries: the fisheries for tunas, swordfish, sharks, and billfish (recreational only) in the U.S. Atlantic, Caribbean, and Gulf of Mexico (GOM) waters, including the pelagic driftnet, drift gillnet, pelagic longline (PLL), bottom longline, purse seine, green stick, and hand gear (rod and reel, bandit gear, handline, buoy gear, and harpoon) fisheries. The focus of this opinion is the PLL component of the Atlantic HMS fisheries (the PLL fishery or the HMS PLL fishery).

Prior Consultations on Atlantic HMS PLL Fishery

Since 1999, consultations on the Atlantic HMS fisheries, including the PLL fishery, have resulted in three biological opinions that are pertinent to the current fishery operations and to the consultation at issue: the June 30, 2000 Opinion, the June 14, 2001 Opinion, and the June 1, 2004 Opinion. A brief history of these consultations is described below. For a more detailed consultation history, please see the discussion in the 2004 Opinion.

On June 30, 2000, NMFS issued an opinion on the effects of the PLL fishery. Among other things, the consultation evaluated proposed regulations (64 FR 69982, Dec. 15, 1999) to reduce bycatch in the PLL fishery. The resulting June 30, 2000 Opinion concluded that the PLL fishery was likely to jeopardize the continued existence of loggerhead and leatherback sea turtles.

To avoid this jeopardy, the June 30, 2000 Opinion offered two possible reasonable and prudent alternatives (RPAs). Shortly after issuing the Opinion, NMFS published emergency regulations to implement the selected RPA temporarily (65 FR 60889; Oct. 13, 2000). Under the emergency regulations, an L-shaped portion of the Northeast Distant (NED) statistical reporting area was closed to Atlantic HMS PLL fishing from October 10, 2000, through April 9, 2001, to reduce incidental capture of loggerhead and leatherback sea turtles. The emergency regulations also required the use of dipnets and line-cutters to remove entangling fishing gear and reduce post-release mortality of sea turtles captured in the PLL fishery. An interim final rule adopted and continued the requirement to possess and use dipnets and line-cutters for all vessels in the PLL fishery (66 FR 17370; March 30, 2001).

After issuing the June 30, 2000 Opinion, NMFS concluded that further analyses of observer data and additional population modeling of loggerhead sea turtles were needed to determine more precisely the impact of the PLL fishery on sea turtles. For that reason, NMFS reinitiated consultation, and issued another biological opinion on June 14, 2001.[1] The Opinion represented a comprehensive examination of the effects of all of the fisheries covered under the 1999 HMS

[1] NMFS issued an Opinion on June 8, 2001, but replaced it with a revised Opinion on June 14, 2001, which included minor edits.

FMP and the 1999 Billfish FMP on listed sea turtles in the western Atlantic Ocean. The Opinion concluded that the operation of the PLL fishery was likely to jeopardize the continued existence of loggerhead and leatherback sea turtles. None of the other fishery components, including the Atlantic bottom longline and gillnet fisheries for sharks, were found likely to jeopardize the continued existence of any ESA-listed species.

The June 14, 2001 Opinion specified an RPA for the PLL fishery that would avoid the likelihood of jeopardizing the continued existence of loggerhead and leatherback sea turtles. The RPA included the following elements:

- Closure of the NED area to HMS PLL fishing, effective July 15, 2001;
- A requirement that gangions be placed no closer than twice the average gangion length from the suspending floatlines, effective August 1, 2001 (the gangion placement requirement);
- A requirement that gangion lengths be 110% of the length of the floatline in sets of 100 meters or less in depth, effective August 1, 2001 (the gangion length requirement);
- A requirement for the use of corrodible hooks, effective August 1, 2001; and
- A requirement for additional gear modification or fishing practices prior to reopening the NED based on a new cooperative research program.

The RPA of the June 14, 2001 Opinion also required NMFS to initiate and conduct a cooperative research program, which would develop, modify, and test gear technologies and fishing strategies to: (1) reduce the likelihood of interactions between fishing gear and sea turtles; and (2) dramatically reduce immediate and delayed mortality rates of sea turtles captured in the fisheries. This research program led to the experimental fishery in the NED, discussed below, which led to the development of additional gear and safe handling requirements to reduce sea turtle interactions and mortality.

The June 14, 2001 Opinion also included a term and condition, to minimize the impacts of incidental take, that required NMFS to take action by September 15, 2001 to require all commercial and recreational HMS-permitted vessels to post, inside the wheelhouse, guidelines for the safe handling and release of sea turtles following longline interactions.

On July 13, 2001, NMFS published an emergency rule to put in place the NED closure, gangion placement, and gangion length requirements of the RPA in the June 14, 2001 Opinion. These requirements were applicable to vessels that use pelagic longline gear and that had been issued or were required to have HMS permits (66 FR 36711; July 13, 2001). The emergency rule also required all vessels with, or required to have, HMS permits, regardless of gear type or target species to post the safe handling and gear release procedures inside the wheelhouse, consistent with a term and condition in the June 14, 2001 Opinion. Later, consistent with a modification to that term and condition, NMFS amended the emergency rule so that it applied only to vessels using bottom longline and pelagic longline gears (66 FR 48812; Sept. 24, 2001). These emergency measures were initially in place until January 9, 2002, but were extended through July 8, 2002 (66 FR 64378, Dec. 13, 2001; 67 FR 1688, Jan. 14, 2002).

On July 9, 2002, NMFS published the final rule (67 FR 45393; July 9, 2002) implementing all of the measures identified in the RPA of the June 14, 2001 Opinion, except for the gangion placement requirement. It also implemented the term and condition regarding posting the safe handling and release requirements, as revised to apply only to vessels using bottom longline and pelagic longline gear. The final rule additionally established regulations for the HMS shark gillnet fishery. NMFS did not implement the gangion placement requirement of the RPA of the June 14, 2001 Opinion 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.

From 2001 until 2003, the SEFSC, in coordination and collaboration with the HMS pelagic longline fishery, academic partners, and other NMFS researchers, undertook a series of research activities in the NED to evaluate measures to reduce sea turtle bycatch and bycatch mortality. These studies, collectively known as the NED experiment, evaluated the effectiveness of different fishing techniques and safe-handling techniques at achieving those goals. This experiment contributed to the development of gear and handling requirements that were subsequently evaluated in the June 1, 2004, Opinion, discussed below.

In November 2003, NMFS reinitiated consultation on the PLL fishery because total takes specified in the ITS in the June 14, 2001 Opinion had been exceeded in 2001 (for leatherbacks) and 2002 (for loggerheads and leatherbacks). Take in the fishery was described in a report published by the SEFSC (Garrison 2003a). The reinitiated consultation also considered a proposed rule that would implement new sea turtle bycatch and mortality reduction measures (i.e., hook and bait requirements, gear removal and handling requirements) that were developed following the NED experiment (69 FR 6621, Feb. 11, 2004). Additionally, the consultation considered a proposed rule to implement the 2002 ICCAT swordfish quota recommendations.

As a result of that reinitiation, on June 1, 2004, NMFS issued an Opinion that the long-term operation of the PLL fishery, as proposed, was likely to jeopardize the continued existence of leatherback sea turtles. However, the Opinion stated that the PLL 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.

The RPA included, among other things: maximization of gear removal; a comprehensive outreach program to ensure that fishermen are aware of the safe handling and gear removal requirements; a net mortality rate performance standard to ensure progress in improving sea turtle handling and gear removal; and requirements to improve monitoring and reporting. On improving monitoring and reporting, in addition to addressing observer coverage, the RPA also required observers to record gear removal information, such as information on the hooking location and the amount of gear that was removed. This information allows NMFS to better estimate total sea turtle mortality, including post-release mortality. The Opinion stated that the RPA would also benefit loggerhead sea turtles and that, where those benefits affect 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. Aspects of the RPA were implemented via final rule published July 6, 2004 (69 FR 40734), including requirements for large circle hooks, bait-type specifications, sea turtle bycatch release equipment requirements, and handling and careful release protocols.

In Tables 1.1 and 1.2, we present the levels established in the 2004 Opinion and the number of total incidental takes, post-release mortality rate, and total mortality since 2004 based on the best available data[2].

[2] The SEFSC calculates estimates of total take and total dead-on-retrieval mortality in the HMS PLL fishery, based on observer information on takes and dead-on-retrieval mortality as well as observer coverage. SEFSC extrapolates this information to the entire fishery based on effort information from logbooks. The HMS Management Division uses observer information on hooking location, condition, and gear remaining on observed sea turtle takes, the post-release mortality criteria updated in Ryder et al. (2006), and the mortality tables revised by SEFSC (NMFS 2012d) to estimate a post-release mortality rate. The HMS Management Division applies the post-release mortality rate to the expected non-lethal interactions in the Science Center estimates (i.e., to the individuals that were not dead-on-retrieval) to calculate total expected post-release mortalities in the fishery. In some memos and other documents that the HMS Management Division prepared, total take and total mortality numbers may be slightly different than the SEFSC estimates. This is because the HMS Management Division calculated estimated total takes and total dead-on-retrieval mortality before the SEFSC provided those estimates, in order to timely track take and mortality. All numbers presented in this Opinion are based on the total take numbers and total dead-on-retrieval mortality as calculated by the SEFSC and the post-release mortality rate as calculated by the HMS Management Division. The SEFSC provided their take estimates in annual reports about interactions through the year 2015. For the years 2016-2018, the SEFSC provided take estimates based on the same methodologies, but they did not publish annual reports after 2015.

Table 1.1 Estimated leatherback and loggerhead sea turtle impacts in the Atlantic PLL fishery vs. 2004 Opinion RPA and ITS levels

		Total Incidental Takes (ITS Level)	Post-Release Mortality Rate (RPA Level)	Total Mortality (RPA Level)	Total Mortality Exceedance?
2004-2006	Leatherback	2,125* (1,981)	22.5% (26.2% by Q1 2005; 19.6% by Q1 2006)	505 (548)	No
	Loggerhead	1,569 (1,869)	28.8% (20.2% by Q1 2005; 18.6% by Q1 2006)	452 (438)	Yes
2007-2009	Leatherback	1,167 (1,764)	24.3% (13.1%)	306 (252)	Yes
	Loggerhead	1,557 (1,905)	23.3% (17%)	372 (339)	Yes
2010-2012	Leatherback	1,007 (1,764)	22.1% (13.1%)	226 (252)	No
	Loggerhead	1,464 (1,905)	25.7% (17%)	377 (339)	Yes
2013 - 2015	Leatherback	947 (1,764)	30.1% (13.1%)	288 (252)	Yes
	Loggerhead	882 (1,905)	27.3% (17%)	250 (339)	No
2016 - 2018	Leatherback	753 (1,764)	35.2 %** (13.1%)	270 (252)	Yes
	Loggerhead	294 (1,905)	27.0** (17%)	85 (339)	No

*Over 47% of the estimated takes during 2004-2006 occurred prior to implementation of circle hooks in quarter 3 of 2004.
** 2018 mortality rate estimated using mean of 2016 – 2017.

Table 1.2 Post-Release Mortality Rate in the Atlantic PLL fishery vs. 2004 Opinion RPA levels

	Leatherback		Loggerhead	
	Post-Release Mortality Rate	RPA Level	Post-Release Mortality Rate	RPA Level
2004	26.0	32.8*	34.8	21.8*
2005	15.4	26.2	23.6	20.2
2006	21.9	19.6	24.5	18.6
2007	25.1	13.1	20.9	17.0
2008	23.7	13.1	25.6	17.0
2009	28.3	13.1	23.0	17.0
2010	21.8	13.1	23.2	17.0
2011	22.4	13.1	28.1	17.0
2012	24.2	13.1	25.4	17.0
2013	25.9	13.1	25.4	17.0
2014	34.4	13.1	34.0	17.0
2015	31.9	13.1	25.2	17.0
2016	34.2	13.1	29.2	17.0
2017	37.1	13.1	23.0	17.0

* Note: the RPA levels for 2004 are for Q3 and 4, after the gear requirements came into effect.

On October 2, 2006, NMFS finalized the Consolidated Atlantic HMS FMP (71 FR 58058), which details the management measures for Atlantic HMS fisheries including the Atlantic PLL fishery. NMFS's Atlantic Highly Migratory Species Management Division (HMS Management Division) determined, consistent with the provisions of the June 1, 2004 Opinion, that none of the measures in the 2006 Consolidated Atlantic HMS FMP were expected to alter fishing practices, techniques, or effort or otherwise affect interactions with protected species or habitat and no additional consultation was conducted.

The HMS Management Division continued to monitor take in the fishery and determined that, between 2004-2006, the estimated total incidental takes for leatherback sea turtles exceeded the amount specified in the ITS in the June 1, 2004 Opinion for the 2004-2006 time period. Thus, the HMS Management Division requested reinitiation of ESA Section 7 consultation on December 22, 2006. On August 9, 2007, SERO PRD determined that the 2004 Opinion remained valid, a new opinion was not required, and no changes were needed to amend the 2004 Opinion. The August 9, 2007 memo reasoned that although the total leatherback take level for the first 3-year ITS was exceeded, the total mortality for that time period was below that predicted in the Opinion. As the jeopardy analysis was based on expected mortalities from the fishery, SERO PRD concluded that the ITS exceedance did not alter the jeopardy determination. The memo also explained that approximately 48% of the take level in the ITS for the 2004-2006 time period was reached before the circle hook requirement in the June 1, 2004 Opinion's RPA went into place in the third quarter of 2004. With the circle hook requirement in place, leatherback takes were expected to be on par with, or lower than, the anticipated takes in the June 1, 2004 Opinion. Thus, going forward, the assumptions regarding effects and jeopardy to the species remained valid, and no additional consultation was conducted.

Thereafter, the HMS Management Division continued to monitor compliance with the ITS in the June 1, 2004 Opinion, and consistency with the RPA. Estimates indicated that the leatherback sea turtle post-release mortality rate was exceeded in all RPA implementation periods, and that total mortality was exceeded in the 2007-2009 period. The RPA was designed to avoid jeopardy for leatherback sea turtles. The implemented measures of the RPA also benefited loggerhead sea turtles by minimizing and monitoring the impacts of take. For loggerhead sea turtles, the RPA post-release mortality rate was exceeded in all RPA performance periods, and the total mortality level was exceeded in 2004-2006, 2007-2009, and 2010-2012. The estimated total number of incidental takes (i.e., total captures) for both species were below the level specified in the ITS except for leatherback sea turtles from 2004 to 2006. Leatherback sea turtle take likely exceeded the ITS level in 2004 to 2006 as a result of interactions prior to implementation of circle hook requirements in the third quarter of 2004, as explained above. Total mortality estimates exceeded the RPA levels because the post-release mortality rates were higher than anticipated, despite the number of total takes being below the ITS for all time periods (except for leatherbacks from 2004 to 2006). The HMS Management Division concluded that the new information indicated that the assumptions in the 2004 Opinion as to the post-release mortality rates that could be achieved had not been as anticipated.

Based on this information, on March 31, 2014, the HMS Management Division requested reinitiation of formal consultation with SERO PRD[3]. In addition, on October 30, 2014, the HMS Management Division requested consideration of the effects of the PLL fishery on the Central and Southwest DPS of scalloped hammerhead sharks and threatened coral species. The HMS Management Division made a "no effect" determination for threatened coral species (staghorn coral (*Acropora cervicornis*), elkhorn coral (*Acropora palmate*), pillar coral (*Dendrogyra*

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[3] The HMS Management Division's request estimated total take and total mortalities based on information available at the time. Their estimates were not identical to the estimates later provided by the SEFSC and used in this Opinion (see Table 1.1) because the HMS Management Division did not have complete fishery effort data at that time. However, the HMS Management Division's preliminary estimates led to their request for reinitiation based on an expected exceedance of the ITS.

cylindrus), boulder star coral (*Orbicella annularis)*, mountainous star coral (*Orbicella faveolata)*, star coral (*Orbicella franksi)*, rough cactus coral (*Mycetophyllia ferox)* as well as designated critical habitat for staghorn and elkhorn corals (NMFS 2015).

Due to SERO PRD staffing issues, consultation was delayed until November of 2016, when consultation resumed. Further delays occurred as a result of additional prioritization issues and staffing shortages, as well as the time to gather information and incorporate analysis for additional newly listed species such as Gulf of Mexico Bryde's whale, oceanic whitetip shark, and giant manta ray, which are considered in the consultation.

2.0 Description of the Proposed Action and Action Area

Through this section 7 consultation, the HMS Management Division is consulting over the effects of operation of the PLL fishery for Atlantic HMS on listed species and critical habitat within the action area.

Atlantic HMS fisheries are managed under the dual authority of the Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act) and the Atlantic Tunas Convention Act (ATCA). Under the Magnuson-Stevens Act, fisheries must be managed to prevent overfishing while achieving optimum yield. Under ATCA, the Secretary of Commerce is required to promulgate regulations, as necessary and appropriate, to implement measures adopted by the International Commission for the Conservation of Atlantic Tunas (ICCAT). The HMS regulations are promulgated pursuant to both statutory authorities. This Opinion analyzes the effects of HMS PLL fishing activities carried out under the 2006 Consolidated Atlantic HMS FMP, as amended, and implementing regulations.

A detailed description of the HMS fisheries was included in the 2006 Consolidated Atlantic HMS FMP, as amended, as well as subsequent annual Stock Assessment and Fisheries Evaluation (SAFE) Reports. The HMS PLL fishery is summarized here.

2.1 Description of the PLL Fishery

<u>Description of Species Caught in HMS PLL Fishery</u>
The PLL fishery for Atlantic HMS primarily targets swordfish, yellowfin tuna, and bigeye tuna in various areas and seasons. Secondary target species include dolphin and albacore tuna. PLL gear also interacts with sharks, many species of which are prohibited from retention (see, e.g., 50 C.F.R. 635.24(a)(5), (9), (10)) and some of which may be landed under specified circumstances. Although PLL gear can be modified (e.g., depth of set, hook type, hook size, bait, etc.) to target swordfish or tunas, it is generally a multi-species fishery. PLL vessel operators are opportunistic, switching gear style and making subtle changes to target the best available economic opportunity on each individual trip. PLL gear sometimes attracts and hooks non-target finfish with little or no commercial value as well as species, such as billfish, that cannot legally be retained by commercial fishermen. PLL gear may also interact with protected species such as marine mammals, sea turtles, and seabirds. Thus, this gear has been classified as a Category I fishery with respect to the Marine Mammal Protection Act (MMPA), based upon the level of mortality and serious injury of marine mammals that occurs incidental to the fishery (84 FR 22051, May 16, 2019). Any species that cannot be landed due to fishery regulations is required to be released, regardless of whether the catch is dead or alive.

<u>Description of HMS PLL Gear</u>
PLL gear is composed of several parts. The primary fishing line, or mainline of the longline system, can vary from 5 to 40 miles in length, with approximately 20 to 30 hooks per mile. Longline gear (Figure 2.1) is set horizontally and can be anchored, floating, or attached to a vessel. Under the HMS regulations (50 CFR § 635.2), a vessel is considered to have pelagic longline gear on board when the following equipment is on board:

1. A power-operated longline hauler;
2. A mainline;

3. Floats capable of supporting the mainline; and
4. Leaders (gangions) with hooks.

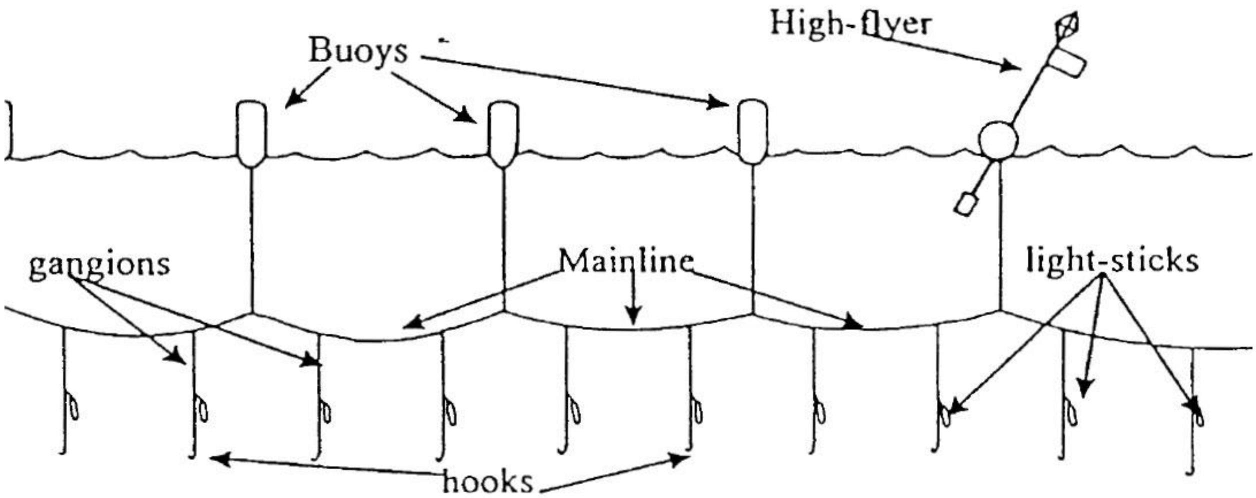

Figure 2.1. Illustration of typical HMS PLL gear. Source: Arocha, 1997 in SAFE 2015

The depth of the mainline can be determined by ocean currents and the length of the floatline, which connects the mainline to several buoys, and periodic markers, which can have radar reflectors or radio beacons attached. Each individual hook is be connected by a leader, or gangion, to the mainline. Lightsticks, which contain light-emitting chemicals, are often used, particularly when targeting swordfish. When attached to the hook and suspended at a certain depth, lightsticks attract baitfish, which may, in turn, attract pelagic predators (NMFS 1999).

When targeting swordfish, PLL gear is generally deployed at sunset and hauled at sunrise to take advantage of swordfish's nocturnal near-surface feeding habits. Except for vessels of the distant water fleet, which undertake extended trips, fishing vessels preferentially target swordfish during periods when the moon is full to take advantage of increased densities of pelagic species near the surface. Typical (shallow set) swordfish gear has floats every 1,000 feet and have 4-5 hooks deployed at a depth of 70-100 feet for each 1,000-foot segment. Mixed target species sets employ a gear arrangement similar to swordfish gear. The number of hooks per set vary with line configuration and target species.

In contrast to swordfish longline gear, longlines targeting tunas are generally set in the morning and hauled back in the evening, use floats much farther apart, have more hooks between the floats, and have the hooks set much deeper in the water column. For example, a typical tuna set can have floats every 1/2 nm, 20-40 hooks on each segment, and hooks set at a depth of 300-1,200 feet. In addition, tuna sets use bait only, while swordfish sets use a combination of bait and lightsticks. Compared with vessels targeting swordfish or mixed species, vessels specifically targeting tuna are typically smaller and fish different areas of water.

Vessel transit to and from the fishing areas is considered part of the proposed action.

Regulations implementing the 2006 Consolidated HMS FMP, as amended, include requirements for vessels fishing with pelagic longline gear. To harvest swordfish and tunas with pelagic longline gear, such vessels are required to have an Atlantic tunas Longline category limited access permit (LAP) holders are required to have), as well as a swordfish LAP (other than handgear or general commercial permit) and a shark LAP, and to fish in accordance with all applicable regulations. (50 CFR § 635.4(d)(4), 635.19(b)). Similarly, a directed or incidental swordfish LAP is valid only when the vessel has on board a valid shark LAP and a valid Atlantic Tunas Longline category LAP issued for such vessel. (50 CFR 635.4(f)(4)). Thus, in the multi-species PLL fishery, all three permits are required in order to fish for authorized Atlantic highly migratory species using pelagic longline gear and to appropriately account for shark bycatch. Regulations implementing the 2006 Consolidated HMS FMP, as amended, include requirements for vessels fishing with pelagic longline gear. Vessels fishing with pelagic longline gear must follow all applicable regulations, which include the following restrictions:

HOOK AND BAIT RESTRICTIONS WHEN FISHING OUTSIDE THE NED

Pelagic longline vessels that are permitted to fish for tunas and swordfish are limited, at all times, to:

- Possessing and/or using only corrodible (i.e., non-stainless steel) 18/0 or larger circle hooks with an offset not to exceed 10 degrees, or 16/0 or larger non-offset circle hooks (50 CFR § 635.21(b)(4); 50 CFR § 635.21(c)(5)(iii)(C))
- Using only whole finfish and/or squid bait (50 CFR § 635.21(c)(5)(iii)(B)).

If green-stick gear is also onboard, a vessel may:

- Possess up to 20 J-hooks 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)(5)(iii)(C)(*4*)).
- J-hooks may be used only with green-stick gear, and no more than 10 hooks may be used at one time with green-stick gear (50 CFR § 635.21(c)(5)(iii)(C)(*4*)).

HOOK AND BAIT RESTRICTIONS WHEN FISHING IN THE NED

In addition to other restrictions, when fishing in the NED Gear Restricted Area, pelagic longline vessels are limited to:

- Possessing onboard and/or using only corrodible (i.e. non-stainless steel) 18/0 or larger circle hooks with an offset not to exceed 10 degrees (50 CFR § 635.21(b)(4); 50 CFR § 635.21(c)(2)(iv)(A)).
- Only whole Atlantic mackerel and/or squid baits may be possessed and/or utilized with the allowed hooks (50 CFR § 635.21(c)(2)(iv)(B)).
- If green-stick gear is also onboard, the same limits on possession and use of J-hooks as noted above apply (50 CFR § 635.21(c)(2)(iv)(A)).

HOOK AND BAIT RESTRICTIONS WHEN FISHING IN THE GULF OF MEXICO

In addition to other restrictions, vessels in the Gulf of Mexico with pelagic longline gear onboard:

- May only possess, use, or deploy circle hooks that are constructed of round wire stock that is no larger than 3.65 mm in diameter (“weak hooks”) between the months of January through June of each calendar year. Two circle hook models that meet this requirement are Mustad Model 39988D – 16/0 and Eagle Claw Model L2048LM – 16/0. Research has shown that white marlin and roundscale spearfish have a higher catch rate on the weak circle hooks compared to the standard circle hooks used elsewhere by the PLL fleet. The seasonal requirement provides protection for large bluefin tuna when they are most prevalent in the GOM, but removing the requirement from July through December when catch rates of white marlin and roundscale spearfish are higher also provides protection to these species (50 CFR § 635.21(c)(5)(iii)(C)(*3*).

- May not use live bait. In addition, no person aboard a vessel with pelagic longline gear onboard may maintain live baitfish in any tank or well onboard the vessel, possess live baitfish, or set up or attach an aeration or water circulation device in or to any such take or well onboard the vessel (50 CFR § 635.21(c)(4)).

OTHER GEAR

Vessels permitted in the PLL fishery may also use other gears such as green-stick gear to catch tunas or other HMS. However, these other gears, including green-stick gear, are not considered in this Opinion, as it they are different gear types. The effects of these gear types to fish for Atlantic HMS species was analyzed in the biological opinion for Atlantic HMS non-PLL fisheries, issued January 10, 2020 (NMFS 2020).

Future Changes to the Pelagic Longline Take Reduction Plan

Under the Marine Mammal Protection Act, the Pelagic Longline Take Reduction Team is charged with developing a Pelagic Longline Take Reduction Plan (PLTRP) to reduce take of marine mammals protected under the MMPA. The PLTRP has been in effect since June 2009, and the Pelagic Longline Take Reduction Team is currently evaluating changes. During meetings in 2015 and 2106, the Pelagic Longline Take Reduction Team came to consensus on changes to requirements related to terminal gear (hooks and gangions), as well as mainline length and setting requirements for PLL fishery activity in specific areas. Although the changes have not yet been proposed in the Federal Register, they are expected to publish sometime in May or June 2020, and thus we discuss them here.

The recommendations for changes to terminal gear requirements (PLTRT 2015) are described as follows:

The goal of these requirements is to make terminal hooks the weakest part of the gear. While pelagic longlining in the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), and Northeast Coastal (NEC), the owner and operator of an Atlantic PLL vessel must use monofilament nylon leaders and/or branch lines that all have a diameter of 1.8 mm or larger (certified by the manufacturer to at least 300 lb breaking force).

- *While pelagic longlining in the FEC, SAB, MAB, and NEC, the owner and operator of an Atlantic PLL vessel must use only hooks meeting criteria at 50 CFR 635.21 and the following specifications:*
    - *16/0 or 18/0 circle hooks with hook shanks containing round wire that can be measured with a caliper or other appropriate gauge, with a wire diameter not to exceed 4.05 mm if 16/0 or 4.4 mm if 18/0; and*
    - *No more than 300 lbs straightening force based on manufacturer's specifications. Hooks that currently meeting these specifications include: 16/0 Mustad 39960D, 16/0 L- 2048-LM Eagle Claw, 16/0 Mustad 39988D, and experimental Lindgren Pitman 18/0 with no offset.*

As stated in the recommendations, these changes are intended to ensure that the hooks are the weakest part of the gear, so that foul-hooked marine mammals such as short-finned pilot whales are more likely to pull free by straightening the hook instead of breaking the gangion and escaping with trailing gear that can cause subsequent problems. The requirements would only apply to the fishery reporting areas along the Atlantic coast (FEC, SAB, MAB, and NEC). These changes are not expected to impact any of the listed species analyzed in this Opinion as they do not alter the likelihood of interaction or nature of interaction for any of those species.

The recommendations for changes to the mainline length and setting requirements (PLTRT 2016) are:

While pelagic longlining in the MAB:

- *An owner and operator of an Atlantic PLL vessel may set no more than 30 nm of active gear (gear with leaders and hooks) with a maximum mainline length of 32 nm, and continuous active gear (gear with leaders and hooks) of no more than 20 nm.*
- *Any active gear in excess of 20 nm must be separated from other active gear along the mainline by a gap of at least 1 nm along the mainline in which no leaders and hooks are set.*
- *There may be no more than one piece of mainline in the water at once (with some exception for line that may become accidentally parted after setting).*

These changes are not expected to increase total effort or alter the likelihood or nature of interactions for any of the species analyzed in the Opinion. These changes only apply to the MAB reporting area. Current PLTRP requirements limit operators to 20 nm mainlines, but do not limit number of sets or total length deployed. Fishermen had been deploying multiple sets, which were found to result in greater pilot whale interactions, compared to the same total mainline length in single sets. The proposed changes are based on pilot whale behavior and how they interact with the fishery.

Fishery Participation and Fishing Effort

This section provides information on past fishery participation and fishing effort to provide context for estimating future effort.

Past Effort
As of October 2019, approximately 280 Atlantic tunas Longline category LAPs had been issued. In addition, approximately 183 directed swordfish LAPs, 71 incidental swordfish LAPs, 218 directed shark LAPs, and 263 incidental shark LAPs had been issued. Atlantic tunas Longline category LAP holders are required to have a swordfish LAP (other than handgear or general commercial permit) and a shark LAP, and to fish in accordance with all applicable regulations (50 CFR § 635.4(d)(4), 635.19(b)). Similarly, a directed or incidental swordfish LAP is valid only when the vessel has on board a valid shark LAP and a valid Atlantic Tunas Longline category LAP issued for such vessel (50 CFR 635.4(f)(4)). Thus, in the multi-species PLL fishery, all three permits are required in order to fish for authorized Atlantic highly migratory species using pelagic longline gear and to appropriately account for shark bycatch.[4] Table 2.1 shows the number of total sets conducted in the PLL fishery from 2005-2018.

Table 2.1 Reported Effort in Pelagic Longline Sets

Year	Total # of Sets
2018	5,635
2017	7,305
2016	6,885
2015	8,195
2014	9,930
2013	10,520
2012	10,539
2011	8,163
2010	7,547
2009	9,346
2008	8,838
2007	8,810
2006	7,697
2005	7,899
Total	**117,309**
Annual Average	**8,379**

Source: Fishery Logbook System, 2005-2018

The Deepwater Horizon Oceanic Fish Restoration Project is a partnership between NOAA, the National Fish and Wildlife Foundation, and pelagic longline fishermen with the purpose of restoring fishery resources damaged in the Gulf of Mexico by the Deepwater Horizon oil

[4] The current number of directed and incidental swordfish LAPs (183 + 71 = 254) is less than the number of permits in the Atlantic tunas longline category LAP (280), even though one of the former permits is a prerequisite to getting the Atlantic tunas longline category LAP. Permits in the swordfish categories terminate if not renewed within one year of expiration, while the Atlantic tunas longline permits do not. The discrepancy results from inclusion of expired but non-renewed permits in the Atlantic tunas longline fishery. Renewal is required to fish with such an Atlantic tunas longline fishery permit.

spill. During the Project, pelagic longline fishermen voluntarily agree to not fish with pelagic longline gear from January to June of each year of participation. This project began in 2017 and will run for approximately six to ten years depending on participation. From 2017 to 2019, seven to ten pelagic longline vessels that normally would have fished with pelagic longline from January to June of each year did not do so, and instead fished with alternative gears such as greenstick, buoy gear, and rod and reel. This change in gear type represents a reduction in pelagic longline effort in the Gulf of Mexico area during the first half of each year of the Project. This project is a temporary project, yet could contribute to a reduction in number of PLL sets beginning in 2017. Participating fishermen are anticipated to return to PLL fishing after their participation in the Project ends. Thus, the reductions in PLL effort resulting from this Project are not anticipated to continue past the remaining years of the Project. However, the Project could have localized effects on PLL fishing effort in the Gulf of Mexico area during the remaining years that the project occurs, depending on levels of participation.

Future Expected Effort
NMFS expects that approximately the same number of permits will be issued in future years as reported in the SAFE (2019) report (see above). Although total effort is variable year to year (Tables 2.1), and therefore difficult to predict with precision for future years, NMFS expects fishing effort in the future will continue to fluctuate around the same levels it has since 2005 since effort has been fairly constant, and no new limited access permits are being issued. NMFS recognizes that effort may increase in the Gulf of Mexico at the end of Deepwater Horizon Oceanic Fish Restoration Project (i.e., sometime between 2022 and 2026), however any potential increase is not known at this time. In addition, averaging effort over the 14 years, from 2005 through 2018, allows us to consider periods of higher effort, including before the program was in place, and lower effort ,and provides the best available information to estimate future effort given the variability over time.

The regulatory management structure of the pelagic longline fishery significantly changed in 2015 with the implementation of Amendment 7 to the 2006 Consolidated HMS FMP. However, this management structure change did not substantially change effort overall or by area in the fishery. The implementation of Amendment 7 shifted the focus of limiting incidental bluefin catch in the HMS pelagic longline fishery from fleet-wide management measures to individual vessel accountability through the implementation of a bluefin tuna catch share program (i.e., the Individual Bluefin Quota (IBQ) Program). Longline category permit holders who qualified for IBQ shares (a percentage of the Longline category quota) through the process established in Amendment 7 annually receive an IBQ allocation (pounds of quota based on the share percentage), which they are required to use to account for incidentally caught bluefin tuna during pelagic longline fishing targeting other tunas and swordfish. Furthermore, fishery participants receive IBQ shares that are designated as either “Gulf of Mexico” or “Atlantic” regional shares. The IBQ Program was designed to reduce bluefin tuna discards within the PLL fishery and to ensure that the bluefin tuna stock continues to be managed within its science-based quota and consistent with conservation requirements.

More recently, NMFS further modified management measures for the Atlantic HMS PLL fishery. On April 2, 2020, NMFS finalized a rule that removes the Cape Hatteras Gear Restricted Area established in Amendment 7, opens two areas previously closed to PLL fishing subject to a monitoring period, and changes a year-round weak hook requirement in the Gulf of Mexico to a seasonal requirement. Implementation of this rule, referred to as the Gear Restricted Area (GRA)/Weak Hook Rule (85 FR 18812), would not change the percentages of IBQ allocation designated as either "Atlantic" or "Gulf of Mexico." Each of these measures takes place within the previously analyzed, science-based quotas for bluefin tuna and consistent with the requirements of the IBQ Program, including IBQ allocation provisions and regional designations. The rule is also not expected to change the amount of effort exerted by pelagic longline fisheries in the Atlantic or Gulf of Mexico areas due to other regulatory restrictions in place for the fishery (i.e., permits are limited access, IBQ allocation costs and availability may have the effect of constraining effort). The rule may change the time and place in which fishing could occur by opening previously closed areas, and it is likely that effort by some participants will be redistributed to the areas that the rule opened to fishing. However, none of these measures are expected increase interactions with endangered or threatened species or critical habitat because the overall effort in the Atlantic or Gulf of Mexico is expected to stay the same and because the distribution of listed species in the opened areas is likely to be the same as the areas where fishing has been occurring. Likewise, there is no information to indicate that implementation of weak hooks would affect the nature of the interactions with protected species analyzed in this Opinion. Therefore, when estimating future interactions based on past fishing effort in this Opinion (Section 5), we do not expect that effort redistribution from the GRA/Weak Hook Rule will result differences in estimated interactions with protected species or in impacts to critical habitat moving forward.

Exempted Fishing, Scientific Research, and Exempted Educational Activity Involving the HMS PLL Fishery

Consistent with regulations at 50 CFR 600.745 and 50 CFR 635.32, NMFS may authorize activities otherwise prohibited by the regulations for the conduct of scientific research, the acquisition of information and data, the enhancement of safety at sea, the purpose of collecting animals for public education or display, the investigation of bycatch, economic discard and regulatory discard, or for chartering arrangements. These activities include, but are not limited to: scientific research resulting in, or likely to result in, the take, harvest, or incidental mortality of Atlantic HMS, including tunas, swordfish, billfish, and sharks from Federal waters in the Atlantic Ocean, Caribbean Sea, and Gulf of Mexico; exempted fishing and educational activities; programs under which regulated species retained in contravention to otherwise applicable regulations may be donated through approved food bank networks; or chartering arrangements.

NMFS issues EFPs, SRPs, and display permits to individuals conducting activities that require exemptions from fishing regulations, for example seasonal or area restrictions or other harvest restrictions. NMFS annually announces its intent to issue exempted fishing permits (EFPs), scientific research permits (SRPs), display permits (See, e.g., 84 FR 64277 (Nov. 21, 2019)). 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 that are regulated under the authority of the Atlantic Tunas Convention Act. 50 CFR 653.32(b).

EFPs, SRPs, and display permits often involve fishing by commercial or research vessels that is similar or identical to the fishing methods of the HMS PLL fishery, the subject of this Opinion.

We consider EFPs, SRPs, and display permits involving fishing consistent with the description of HMS PLL fishing unlikely to increase fishing effort significantly enough to warrant separate consideration in this Opinion. The types and rates of interactions with listed species from these types of EFP, SRP, and display permit activities are expected to be similar to (and fall within) the level of effort and impacts analyzed in this Opinion. For example, issuing an EFP to an active commercial vessel to fish in the same general area and to retain non-ESA-listed species otherwise discarded as regulatory bycatch would not likely result in effects other than those that would result from the vessel's normal commercial activities. Similarly, issuing an EFP or SRP to a vessel to conduct a minimal number of HMS PLL trips would not likely increase fishing effort to a degree that would affect the total annual effort expended in the fisheries, and HMS accounts for any species caught within the relevant quotas. Thus, issuance of EFPs, SRPs, or display permits involving fishing consistent with the description of HMS PLL fishing in this consultation are considered covered by this consultation, and any takes of sea turtles and any other ESA-listed species would be included under the authorized take levels in the incidental take statement of this Opinion.

To ensure consistency with the PLL fishing evaluated under this Opinion (and ensure coverage under this Opinion), the Atlantic Highly Migratory Species Management Division should specify permit conditions similar to the requirements under which the HMS pelagic longline fishery operates to minimize ESA-listed species bycatch and bycatch mortality from fishing activities (e.g., hook type, handling and release equipment), and otherwise ensure consistency with the incidental take statement in this Opinion. If in doubt whether a particular EFP, SRP, or display permit is consistent with this consultation, the HMS Management Division should seek the concurrence of SERO PRD. For EFPs, SRPs, and display permits that are not covered under this consultation, separate consultation, pursuant to section 7 of the ESA, may be required prior to issuance of the permits.

<u>Pelagic Observer Program</u>

The SEFSC Miami Laboratory has been responsible for the administration of the Pelagic Observer Program (POP) since 1992. NMFS places observers aboard HMS-permitted vessels under the authority of the MSA and ATCA, as well as the MMPA and ESA. The objective and mission of the POP is to document the effort, directed catch, and bycatch, as well as collect data on species morphometrics and biological characteristics. Additionally, the program documents fishery interactions with marine mammals, sea turtles, and birds. The observer data are used to estimate catch of target species, bycatch of non-target species, and the incidental take of protected species. Bycatch rates of protected species (catch per 1,000 hooks) are quantified based upon observer data by year, fishing area, and quarter (Garrison 2019). The estimated bycatch rate is then multiplied by the fishing effort (number of hooks) in each area over each quarter and reported to the fishery logbook system (FLS) program to obtain estimates of total interactions for each species of marine mammal and sea turtle (Garrison 2019).

In 2018, NMFS observers recorded 731 PLL sets, an overall fishery coverage of 14.0 % (SAFE 2019). It is anticipated that observer coverage will continue around the average level and will continue to meet the 8 % that was required as a Reasonable and Prudent Alternative in the June 1, 2004 Opinion, and is included as a term and condition in the incidental take statement below.

Actions to Reduce Impacts of Proposed Action

The PLL fishery is subject to management measures designed to help reduce the potential effects of fishing on protected species. They include:

a. Fishermen are required to possess and use protected species safe handling and release gear in compliance with NMFS' careful release protocols. Fishermen are required to attend a training and certification program to ensure that the owner and operator of each permitted HMS vessel authorized to fish with pelagic longline gear knows how to use the protected species safe handling and release gears (50 CFR § 635.21(b)(1) and 50 CFR § 635.8(a)).
b. HMS PLL vessels are required to place the following materials inside the wheelhouse: Technical Memoranda (NMFS-SEFSC-735 and NMFS-SEFSC-738) titled "Careful Release Protocols for Sea Turtle Release with Minimal Injury," (NMFS-SEFSC-735) and "Design Standards and Equipment for Careful Release of Sea Turtles caught in Hook and Line Fisheries," (NMFS-SEFSC 738), and a placard titled "Handling/Release Guidelines" (50 CFR § 635.21(b)(1); 50 CFR § 635.21(c)(2)(iv)(C)).
c. When a marine mammal or sea turtle is hooked or entangled by pelagic longline gear, the operator of the vessel must immediately release the animal, retrieve the pelagic longline gear, and move at least 1 nm (2 km) from the location of the incident before resuming fishing (50 CFR § 635.21(b)(3)).
d. Fishermen must comply with regulations regarding gear and bait requirements (e.g., corrodible circle hooks, etc. as the gear is described in this section) designed to reduce interactions and post-release mortality (50 CFR § 635.21(c)(5)(iii)(B) and (C); 50 CFR § 635.21(b)(4); and 50 CFR 635.21(c)(2)(iv)(A) and (B)).
e. Fishermen must follow protected species resuscitation requirements outlined in the regulations for sea turtles (TM-735) (50 CFR § 635.21 (c)(2)(iv)(D) - (G); 50 CFR § 635.21(c)(5)(i) and (ii)).
f. NMFS ensures that all vessel owners and operators are aware of the potential presence of protected species and the need to avoid collisions while transiting to and from fishing areas, and the need to observe for the presence of protected species to avoid interaction with them. Federal law prohibits approaching or remaining within 500 yd of a North Atlantic Right Whale (50 CFR § 224.103(c)).
g. If the total length of any gangion plus the length of any floatline is less than 100 meters, then the length of all gangions must be at least 10 percent longer than the length of the floatlines (50 CFR § 635.21(c)(5)(iii)(A)).
h. Fishermen cannot deploy a pelagic longline that exceeds 20 NM in length in the mid-Atlantic Bight, with limited exceptions to support research on reducing bycatch of marine mammals in the pelagic longline fishery (PLL Take Reduction Plan Requirement under MMPA) (50 CFR § 229.36(e). Note that this requirement would change if the PLTRT recommendations detailed above are implemented. As explained above, the changes to this requirement would not be expected to alter the analyses and conclusions of this Opinion).
i. Fishermen cannot possess, retain, transship, land, store, or sell certain sharks, including oceanic whitetip and scalloped hammerhead (consistent with ICCAT recommendations) (50 CFR § 635.21(c)(1)(ii)).

2.2 Description of the Action Area

The action area for an Opinion is defined as all of the areas affected by the federal action, 50 CFR 402.02. The HMS PLL fishery operates in large areas of the Gulf of Mexico, the Caribbean Sea, and the Atlantic Ocean, ranging throughout the U.S. EEZ and beyond. Figure 2.2 shows the statistical reporting areas for the HMS PLL fishery and illustrates the wide-ranging nature of the HMS PLL fishery throughout the western North Atlantic Ocean. Figure 2.3 shows areas closed to HMS PLL fishing, including seasonal and year-round closures, and areas with additional gear restrictions for HMS PLL fishing. .

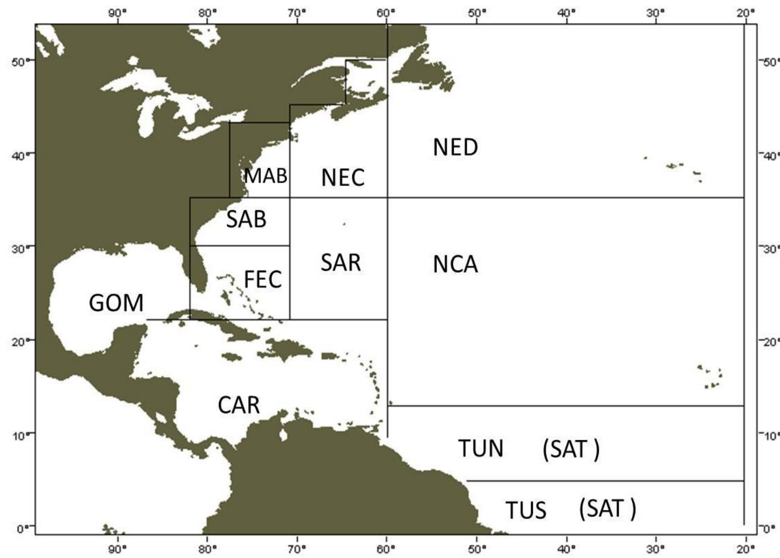

Figure 2.2. HMS PLL fishery reporting areas (CAR: Caribbean; GOM: Gulf of Mexico; FEC: Florida East Coast; SAB: South Atlantic Bight; MAB: Mid Atlantic Bight; NEC: Northeast Coastal; NED: Northeast Distant; SAR: Sargasso; NCA: North Central Atlantic; TUN: Tuna North; TUS: Tuna South). Source: Cramer and Adams 2000.

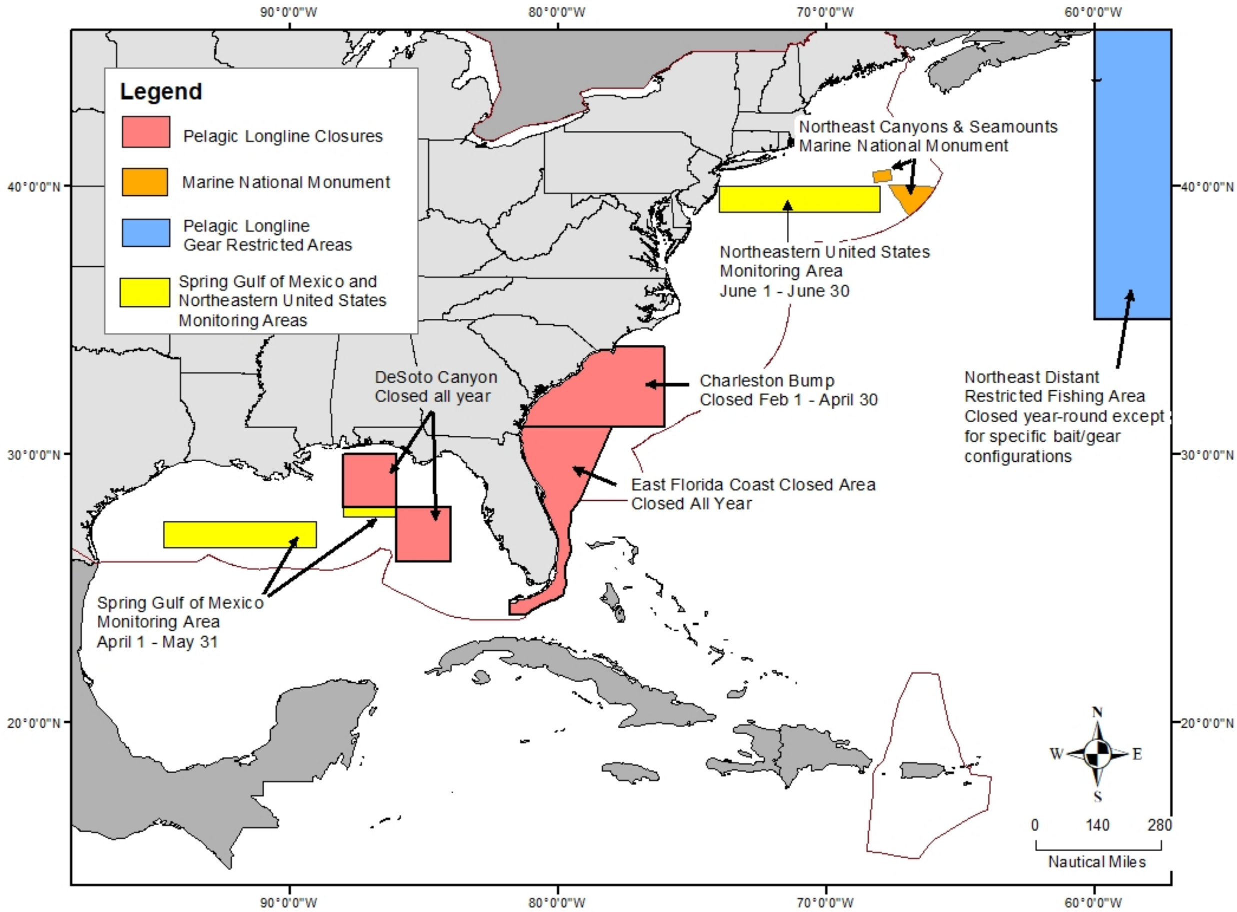

Figure 2.3. Areas Closed/Restricted to HMS Pelagic Longline Fishing by U.S. Flagged Vessels

3.0 Status of the Species and Critical Habitat

Listed species or species proposed for listing occurring within the action area that may be affected by the proposed action include species of whales, sea turtles, and fish. Table 3.1 lists each species, scientific name and status. Designated critical habitat in the action area is listed in Table 3.2.

Table 3.1 Status of Listed Species that May Be Affected in the Action Area (E= endangered, T=threatened)

Species		Scientific Name	Status
Whales	Sei whale	*Balaenoptera borealis*	E
	Blue whale	*Balaenoptera musculus*	E
	Fin whale	*Balaenoptera physalus*	E
	North Atlantic right whale	*Eubalaena glacialis*	E
	Sperm whale	*Physeter macrocephalus*	E
	Gulf of Mexico Bryde's whale	*Balaenoptera edeni*	E
Sea Turtles	Loggerhead sea turtle, Northwest Atlantic (NWA) Distinct Population Segment (DPS)	*Caretta caretta*	T
	Green sea turtle, North Atlantic DPS	*Chelonia mydas*	T
	Green sea turtle, South Atlantic DPS	*Chelonia mydas*	T
	Leatherback sea turtle	*Dermochelys coriacea*	E
	Hawksbill sea turtle	*Eretmochelys imbricata*	E
	Kemp's ridley sea turtle	*Lepidochelys kempii*	E
	Olive ridley sea turtle	*Lepidochelys olivacea*	T
Fish	Atlantic sturgeon, South Atlantic DPS	*Acipenser oxyrinchus oxyrinchus*	E
	Atlantic sturgeon, Carolina DPS	*Acipenser oxyrinchus oxyrinchus*	E
	Atlantic sturgeon, Chesapeake Bay DPS	*Acipenser oxyrinchus oxyrinchus*	E
	Atlantic sturgeon, New York Bight DPS	*Acipenser oxyrinchus oxyrinchus*	E
	Atlantic sturgeon, Gulf of Maine DPS	*Acipenser oxyrinchus oxyrinchus*	T
	Atlantic salmon, Gulf of Maine DPS	*Salmo salar*	E
	Giant manta ray	*Mobula birostris*	T
	Scalloped hammerhead shark, Central and Southwest Atlantic DPS	*Sphyrna lewini*	T
	Smalltooth sawfish	*Pristis pectinata*	E
	Oceanic whitetip shark	*Carcharhinus longimanus*	T

Table 3.2 Designated critical habitat in the action area

Species
North Atlantic right whale
Loggerhead sea turtle: NWA DPS

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

We have determined that the proposed action being considered in this Opinion is not likely to adversely affect the following ESA-listed species: blue whale, sei whale, fin whale, North Atlantic right whale, Gulf of Mexico Bryde's whale, any Atlantic sturgeon DPS, the Atlantic salmon Gulf of Maine DPS, and the smalltooth sawfish U.S. DPS. We also determined that the proposed action will not affect critical habitat for the North Atlantic right whale and the Northwest Atlantic (NWA) DPS of loggerhead sea turtle. The following discussion summarizes our rationale for these determinations. Thereafter, these species and critical habitats are excluded from further analysis and consideration in this Opinion.

3.1.1 Whales

Potential routes of effects to blue, sei, fin, sperm, North Atlantic right, or Gulf of Mexico Bryde's whales from the proposed action include entanglement in fishing gear and collision with HMS PLL 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 ESA-listed whale 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, 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 small portion of marine vessel activity, and that HMS PLL fishing vessels represent an even smaller portion of marine vessel activity; and (3) that despite observer coverage in the fishery since the 1990s, no PLL vessel interactions with large whales have been reported, it is 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 fishing vessel interactions under the proposed action. For the remainder of section 3.1.1, we only analyze potential effects to blue, sei, fin, North Atlantic right whales, and Gulf of Mexico Bryde's whales from pelagic longline gear. Effects to sperm whales from fishing gear interactions are discussed in Section 5.1, below.

Blue and sei whales are predominantly found in offshore waters where pelagic longline fishing targeting HMS occurs, and thus it is possible that they may interact with pelagic longline fishing gear. However, observed or reported interactions between baleen whale species and pelagic longline gear are very rare. No interactions with ESA-listed baleen whales, including blue and sei whales, have been reported, though there have been reports of non-listed minke whales, another baleen whale, interacting with HMS PLL gear in the Atlantic in 2003, 2010, 2013, and 2014 (POP database), with all released alive. In 2003, an unidentified baleen whale was incidentally entangled in pelagic longline gear used in the NED experimental fishery. The fishery observer was unable to definitively identify or photograph the animal. However, it possible that this could have been a blue or sei whale because these species have ranges that overlap with the operation of the fishery and the NED reporting area where the unidentified interaction occurred. The observer was able to document that the animal was released alive with no gear left on the animal. Although the Atlantic Scientific Review Group (ASRG) did not make a “serious injury” determination for this event in accordance with the Marine Mammal Protection Act, based on the serious injury determination criteria for marine mammals (Angliss and DeMaster, 1998), the “unidentified” whale was likely unharmed, with no chance of post release mortality resulting from the interaction. Although it is possible that this interaction was with a blue or sei whale, because there are no recorded interactions with these species over decades of observer coverage, we conclude it is extremely unlikely that this interaction was with a blue or sei whale. For the same reasons (lack of observed interactions), we conclude that gear interactions are extremely unlikely to occur, and thus are not likely to adversely affect these species.

Fin and North Atlantic right whales are more coastal in their distribution, although they can occur in offshore areas as well. Fin and North Atlantic right whales also are baleen whales with ranges that overlap with the operation of the fishery and the NED reporting area where the interaction occurred, and thus the unidentified baleen whale interaction discussed above could have been with one of these species. However, we believe that, because of their more coastal distribution, these whales are even less likely to interact with the longline fishery than the offshore large whales (blue and sei whales). Likewise, there have been no reported or documented interactions between these whale species and the HMS PLL fishery. Given their more coastal distribution, it is extremely unlikely that the 2003 unidentified baleen whale was one of these species. For the same reasons (lack of observed interactions and coastal distribution), we believe that interactions between fin and North Atlantic right whales and PLL fishing gear are extremely unlikely to occur, and gear interactions are not likely to adversely affect these species.

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). Subsequently, a larger “core distribution area” was determined (Figure 3.1).

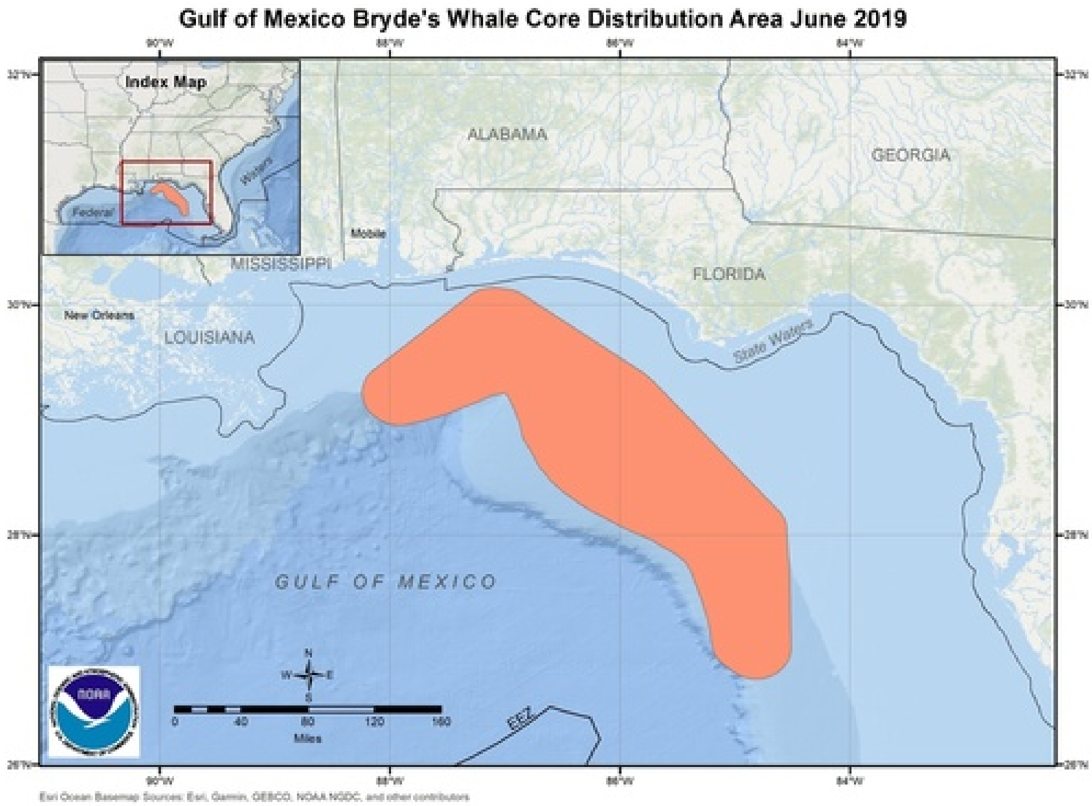

Figure 3.1 (see: https://www.fisheries.noaa.gov/resource/map/gulf-mexico-brydes-whale-core-distribution-area-map-gis-data).

Pelagic longlines are a potential entanglement threat to baleen whales, including the Gulf of Mexico Bryde's whale, since the majority of mainline gear is in the water column (Andersen et al. 2008). The status review for the Gulf of Mexico Bryde's whale cited fishing gear entanglement, including in pelagic longlines, as one of the threats facing the species. Pelagic longline activity in the Bryde's whale core distribution area in the northeastern GOM appears to be lower than in other parts of the GOM. This is likely because much of the area where the Gulf of Mexico Bryde's whale are distributed within the De Soto Canyon Closed Area, which is closed to commercial pelagic longline fishing year round (65 FR 47213, August 1, 2000). A large portion of the broader GOM Bryde's whale's core distribution area overlaps with the De Soto Canyon Closed Area, as does about 2/3 of the narrower BIA, which is where the species would be expected to be the most densely distributed. Approximately 50% of Bryde's whale recorded locations are within the De Soto Canyon Closed Area (Rosel et al. 2016). Outside of the De Soto Canyon Closed Area, however, the fishery has the potential to interact with the species, though there have been no observed interactions with this species and the HMS PLL fishery anywhere within its range, including its core distribution and or the BIA. Of observed sets since 2004 (the year that the most recent fishery changes were implemented), 7.9% (21 of 266) have been in the BIA, with no interactions (SEFSC unpublished data). Due to the lack of recorded interactions, the large proportion of the species' primary range occurring within a marine protected area, the relatively limited effort within the species' primary range, and the rarity of the species, NMFS believes it is extremely unlikely that the HMS PLL fishery would interact with or affect the species.

Because blue, sei, fin, North Atlantic right, and GOM Bryde's whales occur in the action area, we acknowledge there is a possibility of interaction with the pelagic longline fishing gear. The available evidence indicates interactions with baleen whales are exceedingly rare, and typically non-injurious. We believe the chances of a blue, sei, fin, North Atlantic right, or GOM Bryde's whales being adversely affected by the HMS PLL fishing gear are discountable. We conclude the proposed action is not likely to adversely affect these species, and these species will not be considered further in this Opinion.

3.1.2 Atlantic Sturgeon (All DPSs)

Subadult and adult Atlantic sturgeon (all DPSs) live in coastal waters and estuaries when not spawning, generally in shallow (10-50 m depth) nearshore areas dominated by gravel and sand substrates (77 FR 5914, February 6, 2012). Atlantic sturgeon are benthic foragers and prey upon a variety of species in marine and estuarine environment (81 FR 36078, June 16, 2016). In the ocean, Atlantic sturgeon typically occur in waters less than 50 m deep (81 FR 36078, June 16, 2016). The HMS PLL fishery includes longline gear that would be fished in the pelagic environment suspended in the water column. Pelagic hook-and-line gear (i.e., PLL longlines) would not be fished in a manner or depth that would be expected to affect Atlantic sturgeon. Based on this information, it is highly unlikely that the action being considered in this Opinion will affect any Atlantic sturgeon DPS and NMFS considers potential routes of effect from the HMS PLL action to be discountable. We conclude the proposed action is not likely to adversely affect these species, and they will not be considered further in this Opinion.

3.1.3 Gulf of Maine Atlantic Salmon DPS

The endangered Gulf of Maine Atlantic salmon DPS includes the wild population of Atlantic salmon found in rivers and streams from the lower Kennebec River north to the U.S.-Canada border. Atlantic salmon are an anadromous species. Spawning and juvenile rearing occur in freshwater rivers followed by migration to the marine environment. 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. Captures of wild Atlantic salmon in U.S. commercial fishing or by research/survey operations are rare. There have been a few reported taken by trawls in the Gulf of Maine and southern New England, but there are no records since 1992. Based on this information, it is highly unlikely that the action being considered in this Opinion will affect the Gulf of Maine DPS of Atlantic salmon and NMFS considers potential routes of effect from the PLL action to be discountable. We conclude the proposed action is not likely to adversely affect this species, and it will not be considered further in this Opinion.

3.1.3 Smalltooth Sawfish U.S. DPS

Smalltooth sawfish historically occurred commonly in the shallow waters of the Gulf of Mexico and along the eastern seaboard as far north as North Carolina, with rare records of occurrence as far north as New York. The smalltooth sawfish range has subsequently contracted to predominantly peninsular Florida and, within that area, they are found with most regularity off the extreme southern portion of the state. Smalltooth sawfish are generally shallow warm-water fish, known to spend most of their time at or near the bottom of inshore bars, mangrove edges, and seagrass beds. Younger (smaller) animals are believed to be restricted to shallow depths; however, larger animals roam over a much greater depth range, with records from as deep as over 70 m. There have been no documented interactions between the HMS PLL fishery and smalltooth sawfish by NMFS observers, and no other reports of interactions have been found. The only areas where smalltooth sawfish are likely to occur in the U.S. EEZ in the Atlantic are off the coast of Florida and northern Georgia. Since March 1, 2001, the waters off the east coast of Florida have been closed to HMS PLL fishing year-round, and the Charleston Bump, which encompasses federal waters off of Georgia, is closed seasonally to HMS PLL fishing (Feb. 1 to April 30) (See Figure 2.3).

Based on the rarity of smalltooth sawfish in federal waters where HMS PLL fishing occurs, their use of benthic habits (versus the HMS PLL fishery occurring in the water column), and the absence of records in observer data, it is highly unlikely that the action being considered in this Opinion will affect the smalltooth sawfish and potential routes of effect are considered discountable. We conclude the proposed action is not likely to adversely affect this species, and this species will not be considered further in this Opinion.

3.1.4 North Atlantic Right Whale Critical Habitat

NMFS originally designated critical habitat for the North Atlantic right whale in the North Atlantic Ocean when right whales were still recognized as a single species (59 FR 28793, July 5, 1994). Right whales were subsequently designated as two separate species, North Pacific and North Atlantic right whales (73 FR 12024, March 6, 2008) following a status review, but the critical habitat designation from 1994 was maintained. Then on January 27, 2016, NMFS published a Final Rule expanding the critical habitat designation for the North Atlantic right whale (81 FR 4838). Two units were designated for critical habitat, one in the northeastern U.S., the other the southeastern U.S. The boundaries of the critical habitat units are shown in Figure 3.2. The current physical and biological features essential to the conservation of endangered North Atlantic right whales are described here.

Unit 1

Unit 1 provides foraging area functions for the species. The physical and biological features essential to the conservation of the North Atlantic right whale for Unit 1 are the physical oceanographic conditions and structures of the Gulf of Maine and Georges Bank region that combine to distribute and aggregate zooplankton (food) for right whale foraging, namely prevailing currents and circulation patterns, bathymetric features (basins, banks, and channels), oceanic fronts, density gradients, and temperature regimes; low flow velocities in Jordan, Wilkinson, and Georges Basins that allow diapausing zooplankton to aggregate passively below the convective layer so that they are retained in the basins; late stage zooplankton in dense aggregations in the Gulf of Maine and Georges Bank region; and diapausing zooplankton in aggregations in the Gulf of Maine and Georges Bank region.

Unit 2

Unit 2 provides calving habitat for the species. The physical features essential to the conservation of the North Atlantic right whale for Unit 2 are: (1) calm sea surface conditions of Force 4 or less on the Beaufort Wind Scale; (2) sea surface temperatures from a minimum of 7°C, and never more than 17°C; and (3) water depths of 6-28 m, where these features simultaneously co-occur over contiguous areas of at least 231 km^2 of ocean waters during the months of November through April.

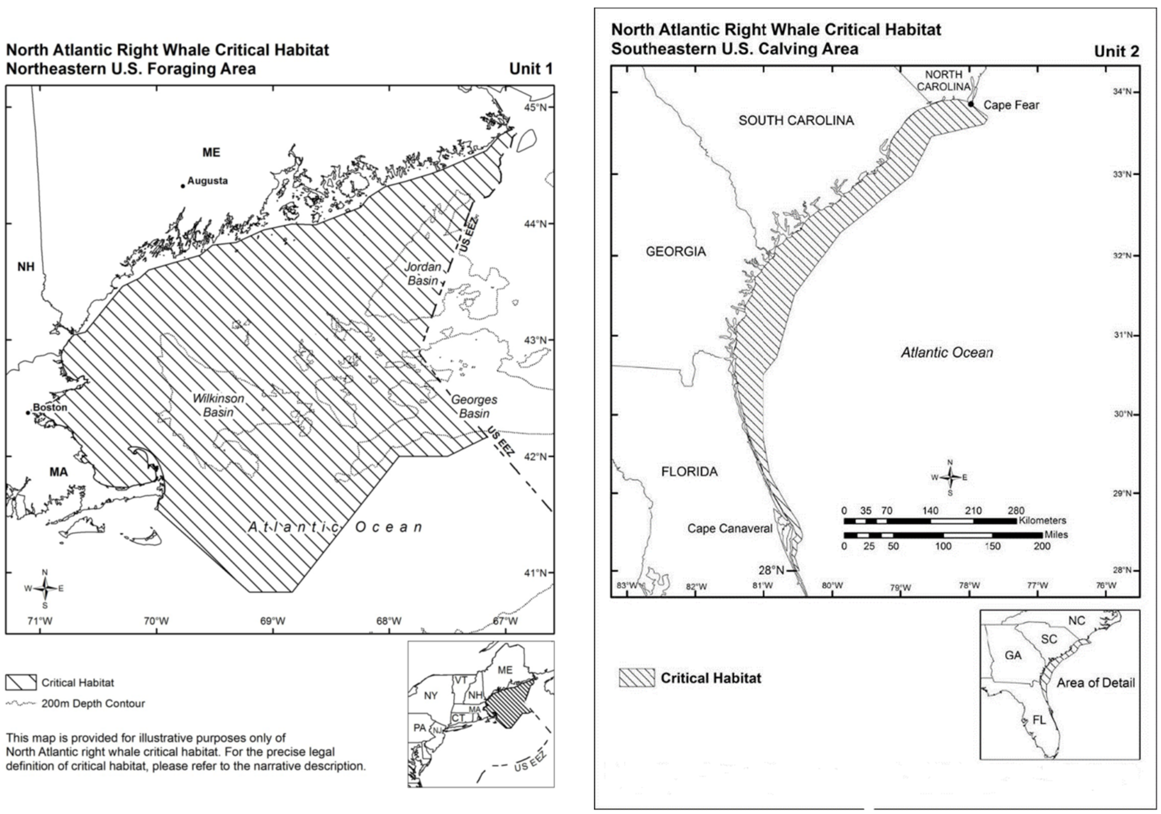

Figure 3.2 North Atlantic right whale critical habitat (Source: 81 FR 4838, January 27, 2016)

None of the gear types/techniques or vessel activities associated with the proposed action would affect the essential features of either of these units. These activities would have no ability to alter physical oceanographic conditions and structures of the Gulf of Maine and Georges Bank region and thus would not alter the distribution and aggregation of zooplankton. Similarly, the proposed action would not alter sea state, sea surface temperature, or water depth, individually or when they co-occur. Thus, the proposed action would not affect designated critical habitat for the North Atlantic right whale and it will not be considered further in this Opinion.

3.1.5 Loggerhead Sea Turtle (NWA DPS) Critical Habitat

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

The HMS Management Division determined that the HMS PLL fishery would have no effect on all critical habitat types, except *Sargassum*. We believe the proposed action would have no effect on any of the habitat types, including *Sargassum*.

The proposed action could potentially occur near or transit through *Sargassum* habitat, however it would not affect *Sargassum* concentrations or community. 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 inhabitance of loggerheads. Likewise, the fishery would not affect *Sargassum* concentrations that support adequate prey abundance and cover. The proposed action would not affect the availability of prey or other material associated with *Sargassum* habitat as the fishery is not targeting or incidentally harvesting smaller prey species or *Sargassum*. Nor does the fishery have the capability to affect the water depth or proximity to currents necessary for offshore transport, or the foraging and cover requirements for post-hatchling loggerheads. While some vessels associated with these fisheries may transit through *Sargassum* habitat, those vessel tracks will not alter *Sargassum* concentrations or otherwise affect its ability to support adequate prey abundance and cover. Further, the wakes and surface water disruption associated with these vessels will not affect the distribution of *Sargassum* mats. We conclude the proposed action will not affect the *Sargassum* habitat, and therefore will not affect loggerhead critical habitat, and it will not be considered further in this Opinion.

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		
Sargassum 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 *Sargassum*	1) Convergence zones, surface-water downwelling areas, and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for optimal growth of *Sargassum* and inhabitance of loggerheads 2) *Sargassum* in concentrations that support adequate prey abundance and cover 3) Available prey and other material associated with *Sargassum* habitat such as, but not limited to, plants and cyanobacteria and animals endemic to the *Sargassum* 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 Sargassum for post-hatchling loggerheads (i.e., >10 m depth to ensure not in surf zone)

3.2 Species Likely to be Adversely Affected

Sperm whale, NWA DPS loggerhead sea turtle, NA DPS green sea turtle, SA DPS green sea turtle, hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, olive ridley sea turtle, giant manta ray, Central and Southwest Atlantic DPS scalloped hammerhead shark, and oceanic whitetip shark are all likely to be adversely affected by the proposed action. These species are highly migratory, travel widely throughout the GOM and Atlantic, and are known to occur in areas subject to PLL fishing. The remaining sections of this Opinion will focus solely on these species.

The following subsections are synopses of the best available information on the status of the species that are likely to be adversely affected by the proposed action, including information on the distribution, population structure, life history, abundance, and population trends of each species and threats to each species. The biology and ecology of these species as well as their status and trends inform the effects analysis for this Opinion. Background information on the sperm whale can be found in the recovery plan (NMFS 2010b) and five year review (NMFS 2015). Additional background information on the status of sea turtle species can be found in a number of published documents, including: recovery plans for the Atlantic green sea turtle (NMFS and USFWS 1991a), hawksbill sea turtle (NMFS and USFWS 1993a), Kemp's ridley sea turtle (NMFS and USFWS 1992c), leatherback sea turtle (NMFS and USFWS 1992b), and loggerhead sea turtle (NMFS and USFWS 2008a); Pacific sea turtle recovery plans (NMFS and USFWS 1998b; NMFS and USFWS 1998d; NMFS and USFWS 1998e; NMFS and USFWS 1998b); and sea turtle status reviews, stock assessments, and biological reports (Conant et al. 2009b; NMFS-SEFSC 2001; NMFS-SEFSC 2009b; NMFS and USFWS 1995; NMFS and USFWS 2007a; NMFS and USFWS 2007c; NMFS and USFWS 2007e; NMFS and USFWS 2007f; NMFS and USFWS 2007h; Seminoff et al. 2015b; TEWG 1998b; TEWG 2000b; TEWG 2007b; TEWG 2009b). The best available information on giant manta ray can be found in the status review report (Miller and Klimovich 2017), the proposed listing rule (82 FR 3694, Jan. 12, 2017), and the final listing rule (83 FR 2916, Jan. 22, 2018). Information regarding the Central and Southwest Atlantic DPS of scalloped hammerhead shark can be found in status review report (Miller et al. 2014), proposed listing rule (78 FR 20717, Apr. 5, 2013), and final listing rule (79 FR 38213, Jul. 3, 2014). Background information on the Oceanic whitetip shark can be found in the status review report (Young et al. 2016), the proposed listing rule (81 FR 96304, Dec. 29, 2016), and the final listing rule (83 FR 4153, Jan. 30, 2018).

3.2.1 Sperm whale

Sperm whales were first listed under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973. The sperm whale is endangered as a result of past commercial whaling. The IWC estimates that nearly 250,000 sperm whales were killed worldwide in whaling activities between 1800 and 1900. From 1910 to 1982, nearly 700,000 sperm whales were killed worldwide by whaling activities (IWC Statistics 1959 to 1983). A compilation of all whaling catches in the North Atlantic north of 20ºN from 1905 onward gave totals of 28,728 males and 9,507 females (NMFS 2010a). Sperm whales are also protected under the MMPA and listed in Appendix I of the Convention on International Trade in Endangered Species of Wild

Fauna and Flora (CITES), meaning that commercial trade in products of sperm whales is prohibited.

Species Description and Distribution
The sperm whale occurs in all oceans of the world. Sperm whales are perhaps the most widely distributed mammal on earth. It is the largest of the toothed whales, reaching a length of 60 feet (18.3 meters) in males and 40 feet (12.2 meters) in females (Odell 1992). Sperm whales are distributed throughout most oceanic areas but are found in deeper waters seaward of the continental shelf. Deep water is required so they can make prolonged, deep dives to locate prey, breed, and nurse their young. In general, females and immature sperm whales appear to be restricted in range, whereas males are found over a wider range and do make occasional movements across and between ocean basins (Dufault et al. 1999). Stable, long-term associations among related and unrelated females form the core units of sperm whale societies (Christal and Whitehead 1998). Females and juveniles form groups that are generally distributed within tropical and temperate latitudes between 50°N and 50°S, while the solitary adult males can be found at higher latitudes between 75ºN and 75ºS (Reeves and Whitehead 1997). The home ranges of individual females seem to span distances of approximately 1,000 kilometers (Best 1979; Dufault and Whitehead 1995). Although there is strong evidence for geographic, matrilineal structuring in sperm whales, there is no evidence the management stocks presented in the following paragraph represent distinct populations of whales.

The Recovery Plan (NMFS 2010a) identifies recovery criteria geographically across three ocean basins: the Atlantic Ocean/Mediterranean Sea, the Pacific Ocean, and the Indian Ocean. This geographic division by basin is due to the wide distribution of sperm whales and presumably little movement of whales between ocean basins. For management purposes under the MMPA, sperm whales inhabiting U.S. waters have been divided into five stocks: (1) the California-Oregon-Washington Stock, (2) the North Pacific (Alaska) Stock, (3) the Hawaii Stock, (4) the Northern Gulf of Mexico Stock, and (5) the North Atlantic Stock. In the Gulf of Mexico, sperm whales are the most common large cetacean seaward of the continental shelf (Davis et al. 1998; Jefferson and Schiro 1997; Mullin et al. 1991; Mullin and Fulling 2004; Mullin et al. 1994; Weller et al. 2000; Wursig et al. 2000). Sperm whales in the Gulf of Mexico are not evenly distributed, showing greater densities in areas associated with oceanic features that provide the best foraging opportunities (**Figure**).

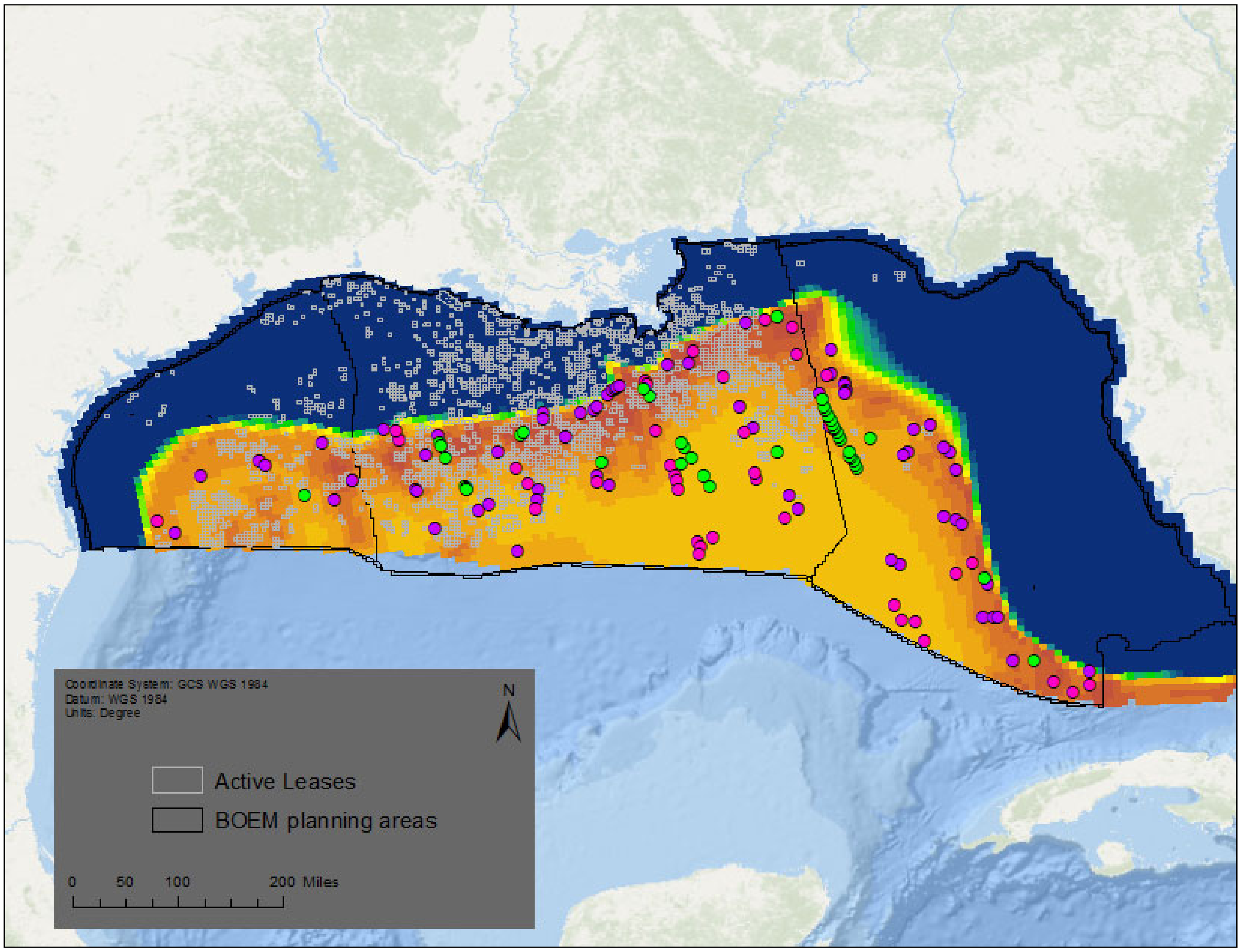

Figure 3.3 Sperm whale sightings (circles with different colors representing different season trips) overlaying Roberts et al. (2016b) mean abundance in the Gulf of Mexico (2003-2004 Southeast Fisheries Science Center Survey Data).

Life History Information
The social organization of sperm whales, as with most other mammals, is characterized by females remaining in the geographic area in which they were born and males dispersing more broadly. Females group together and raise young. For female sperm whales, remaining in the region of birth can include very large oceanic ranges the whales need to successfully forage and nurse young whales. Male sperm whales are mostly solitary and disperse more widely and can mate with multiple female populations throughout a lifetime.

Female and immature sperm whales of both sexes are found in more temperate and tropical waters throughout the year. Maturing males will leave the female groups and form loose aggregations of bachelor schools. As the males grow older, they separate from the bachelor schools and remain solitary most of the year (Best 1979). Adult males visit female groups of whales only to breed. Large males have been sighted on occasion and are believed to enter the Gulf of Mexico for short periods to breed. Therefore, the Gulf of Mexico population is comprised of the year-round presence of females, calves, and juvenile whales. The proportion of females to males in the Gulf of Mexico is 72:28 (Engelhaupt et al. 2009). Calves make up about 11 percent of the population in the Gulf of Mexico (Jochens et al. 2008).

Female sperm whales attain sexual maturity at a mean age of eight or nine years. Mature females ovulate April through August in the Northern Hemisphere. Maturation in males usually begins in this same age interval as females, but males have a prolonged puberty and attain sexual maturity at between age 12 and 20. Males may require another 10 years to become large enough to successfully compete for breeding rights (Kasuya 1991). During this season of ovulating females, one or more large mature bulls temporarily join each breeding school. In the North Atlantic Ocean, the peak breeding season for sperm whales occurs during the spring (March/April to June), although some mating activity continues throughout the summer (NMFS 2015c). In the South Atlantic Ocean, the peak breeding season is presumed to occur in the austral spring. During mating seasons, mature males in their late twenties and older rove among groups of females. Because females within a group often become reproductively active at the same time, the male need not remain with them for an entire season to achieve maximal breeding success (Best and Butterworth 1980) and their association with a female group can be as brief as several hours. Gestation lasts well over a year, with credible estimates of the normal duration ranging from 15 months to over 18 months. A single calf is born at a length of about 13 feet (four meters). Female sperm whales rarely become pregnant after the age of 40 (Whitehead and Mesnick 2003). It is thought that females assist each other in the care of offspring, guarding of young at the surface while mothers dive (Whitehead 1996). Females even have been observed nursing calves other than their own (Reeves and Whitehead 1997). Calves are nursed for two to three years (in some cases, up to 13 years), and the calving interval is estimated to be about four to seven years (Kasuya 1991).

The age distribution of the sperm whale population is unknown, but they are believed to live at least 60 years (Rice 1989). Potential sources of natural mortality in sperm whales include killer whale predation and disease (Lambertsen 1997; Whitt et al. 2015). Sperm whales may also be “harassed” by pilot whales (*Globicephala spp*.) and false killer whales (*Pseudorca crassidens*), but most “attacks” by these species are probably unsuccessful (Palacios and Mate 1996; Weller et al. 1996). Very little is known about the role of disease in the natural mortality of sperm whales (Lambertsen 1997). Several naturally occurring diseases that are likely to be lethal have been identified in sperm whales: myocardial infarction associated with coronary atherosclerosis, gastric ulceration associated with parasitic nematode infection**,** the papilloma virus, (Lambertsen 1997) and *Brucella* and *Morbillivirus* (West et al. 2015). There were 37 individual sperm whale strandings reported in the Gulf of Mexico from 2000-2016 in Texas, Louisiana, Alabama and Florida (NOAA National Marine Mammal Health and Stranding Response Database unpublished data). At least seven of those reported were calves. Using data from 2003-2007, Williams *et al.* (2011) suggested that the rate of recovery of sperm whale carcasses in the Gulf of Mexico was 3.4 percent.

Cephalopods (i.e., squid, octopi, cuttlefishes, and nautili) are the main component of sperm whale diets. The ommastrephids, onychoteuthids, cranchids, and enoploteuthids are the cephalopod families that are numerically important in the diet of sperm whales in the Gulf of Mexico (Davis et al. 2002). Other populations, especially mature males in higher latitudes, are known to feed on significant quantities of large demersal and mesopelagic sharks, skates, and bony fishes (Clarke 1962; Clarke 1979). Sperm whales consume about 3.0 to 3.5 percent of their body weight per day (Lockyer 1981). Sperm whales undergo deep foraging dives to find prey, spending approximately 75 percent of their day in the foraging dive cycle (Watwood et al. 2006). Descent rates are approximately 1.7 meters per second and nearly vertical (Goold and Jones 1995). Dive depth may be dependent upon temporal variations in prey location in the water column. Typical foraging dives last 40 minutes to depths of about 1,300 feet (400 meters), followed by approximately eight minutes of resting at the surface (Gordon 1987; Papastavrou et al. 1989). Nonetheless, dives of over two hours and deeper than 3.3 kilometers (2 miles) have been recorded (Clarke 1976); individuals may spend extended periods of time at the surface to recover.

The highly asymmetrical, disproportionately large head of the sperm whale is an adaptation to produce acoustic signals (Cranford 1992; Norris et al. 1972). Recordings of sperm whale vocalizations reveal that they produce a variety of sounds, such as clicks, gunshots, chirps, creaks, short trumpets, pips, squeals, and clangs (Goold 1999). Sperm whales locate prey by echolocation clicks while in a deep dive pattern, and also produce vocalizations while resting at the surface. The function of vocalizations is relatively well-studied (Goold and Jones 1995; Weilgart and Whitehead 1997). Long series of monotonous, regularly spaced clicks and closely spaced clicks are produced for echolocation and are associated with feeding and prey capture (Goold and Jones 1995; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). However, clicks are also used in short patterns (codas) during social behavior and intragroup interactions (Gero et al. 2015; Gero et al. 2016; Weilgart and Whitehead 1993). Sperm whales show regional differences in coda patterns (Gero et al. 2016; Weilgart and Whitehead 1997). Clicks may also aid in intra-specific communication. Clicks are heard most frequently when sperm whales are engaged in diving and foraging behavior (Miller et al. 2004; Whitehead and Weilgart 1991). Creaks (rapid sets of clicks) are heard most frequently when sperm whales are foraging and engaged in the deepest portion of their dives, with inter-click intervals and source levels being altered during these behaviors (Laplanche et al. 2005; Miller et al. 2004). When sperm whales are socializing, they tend to repeat series of group-distinctive clicks (codas), which follow a precise rhythm and may last for hours (Watkins and Schevill 1977). Codas are shared between individuals in a social unit and are considered to be primarily for intragroup communication (Rendell and Whitehead 2004; Weilgart and Whitehead 1997). Recent research in the South Pacific Ocean suggests that in breeding areas the majority of codas are produced by mature females (Marcoux et al. 2006). Coda repertoires have also been found to vary geographically and are categorized as dialects, similar to those of killer whales (Pavan et al. 2000; Weilgart and Whitehead 1997). For example, significant differences in coda repertoire have been observed between sperm whales in the Caribbean Sea and those in the Pacific Ocean (Weilgart and Whitehead 1997). Three coda types used by male sperm whales have recently been described from data collected over multiple years: these codas associated with dive cycles, socializing, and alarm (Frantzis and Alexiadou 2008).

Our understanding of sperm whale hearing stems largely from the sounds they produce. The only direct measures of sperm whale hearing were conducted on a stranded neonate using the auditory brainstem response technique: the whale showed responses to pulses ranging from 2.5 to 60 kHz and highest sensitivity to frequencies between five to 20 kHz (Ridgway and Carder 2001). Other hearing information consists of indirect data. For example, the anatomy of the sperm whale's inner and middle ear indicates an ability to best hear high-frequency to ultrasonic hearing (Ketten 1992). The sperm whale may also possess better low-frequency than other odontocetes, although not as low as many baleen whales (Ketten 1992). Reactions to anthropogenic sounds can provide indirect evidence of hearing capability, and several studies have made note of changes seen in sperm whale behavior in conjunction with these sounds. For example, sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echo sounders and submarine sonar (Watkins et al. 1985; Watkins and Schevill 1975). In the Caribbean, Watkins et al. (1985) observed that sperm whales exposed to 3.25 to 8.4 kHz pulses (presumed to be from submarine sonar) interrupted their activities and left the area. Similar reactions were observed from artificial sound generated by banging on a boat hull (Watkins et al. 1985). André et al. (1997) reported that foraging whales exposed to a 10 kHz pulsed signals did not ultimately exhibit any general avoidance reactions: when resting at the surface in a compact group, sperm whales initially reacted strongly, and then ignored the signal completely (André et al. 1997). Thode et al. (2007) observed that the acoustic signal from the cavitation of a fishing vessel's propeller (110 dB re: 1 μPa^2 between 250 Hz and 1 kHz) interrupted sperm whale acoustic activity and resulted in the animals converging on the vessel.

A sperm whale was tagged for a controlled exposure experiment during a behavioral response study in southern California and did not appear to demonstrate obvious behavioral changes in dive pattern or production of clicks (Miller et al. 2012; Sivle et al. 2012; Southall et al. 2011).

Clicks produced by sperm whales (and presumably heard by them) are in the range of about 0.1 to 20 kHz (Goold and Jones 1995; Watkins 1977; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997), up to 30 kHz, often with most of the energy in the two to four kHz range (Watkins 1980). Clicks have source levels estimated at 171 dB re: 1 µPa (Levenson 1974). The clicks of neonate sperm whales are very different from typical clicks of adults in that they are of low directionality, long duration, and low frequency (between 300 Hz and 1.7 kHz) with estimated source levels between 140 to 162 dB re: 1 µPa at 1 m (rms) (Madsen et al. 2003).

Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broadband clicks in the frequency range of 100 Hz to 20 kHz that can be extremely loud for a biological source (200 to 236 dB re: 1 µPa at 1 m [rms]), although lower source level energy has been suggested at around 171 dB re: 1 µPa at 1 m (rms) (Goold and Jones 1995; Møhl et al. 2003; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). Another class of sound, "squeals," are produced with frequencies of 100 Hz to 20 kHz (e.g., Weir et al. 2007).

Status and Population Dynamics

The best estimate of the current worldwide abundance of sperm whale is estimated to be between 300,000 and 450,000 individuals (Whitehead 2002) and he abundance of sperm whales in the Atlantic Ocean is estimated at 90,000 to 134,000 individuals (NMFS 2010). There are three stocks of Sperm whales that occur in the action area. The North Atlantic stock has a best estimate of abundance of 2,288 and minimum population estimate of 1,815 based on surveys conducted in 2011 (Waring et al. 2016). The Northern Gulf of Mexico stock which has a best estimate of abundance of 1,436 and a minimum population estimate of 1,164 based on surveys conducted in 2017 and 2018 (Hayes et al. 2020, in review). Previous estimates include 763 resident whales in the northern Gulf of Mexico, according to the 2015 stock assessment report (NMFS 2015c). The pre-spill abundance estimate for sperm whales in the Gulf of Mexico is 1,635 individuals (DWH Trustees 2015). The estimate of 763 individuals is based on an oceanic survey from 2009, whereas the estimate of 1,635 individuals is based upon sighting functions as well as a spatially explicit model of sperm whale density that was used for the injury quantification analysis for the Deepwater Horizon oil spill (DWH MMIQT 2015). Roberts et al. (2016a) used a habitat-based distribution model and estimated 2,128 sperm whales in the Gulf of Mexico. Puerto Rico and U.S. Virgin Islands stock is provisionally considered a separate stock for management purposes, although there is currently limited information to differentiate this stock from the Atlantic Ocean stock(s) and there is insufficient data to determine abundance estimates (Waring et al. 2010).

On a global scale, no genetic differences have been found in the nuclear DNA (nDNA) (bi-parentally inherited) between individuals sampled in different ocean basins with some differences found in mitochondrial DNA (mtDNA) (maternally-inherited) sequences (Lyrholm et al. 1999). In general, results tend to find low genetic differentiation of nDNA among sperm whales in different ocean basins and little differentiation of mtDNA within ocean basin stocks, with the exception of some semi-enclosed basins such as the Mediterranean Sea and Gulf of Mexico (Bond 1999; Engelhaupt 2004; Lyrholm and Gyllensten 1998; Lyrholm et al. 1999; Mesnick et al. 1999; Richard et al. 1996). Based on over 2,473 tissue samples and 1,038 mtDNA sequences from a global consortium of investigators, 28 haplotypes have been identified worldwide, defined by 24 variable sites (Mesnick et al. 2005). Three common haplotypes dominated the sequencing and made up 82 percent of the total. This dominance by a few haplotypes indicates broad reproductive mixing of genetic material. Mitochondrial DNA evidence in the Gulf of Mexico suggests population structuring based on genetic material inherited from mothers. Regional structuring is also supported by satellite tracking data suggesting that most females establish home ranges within the Gulf of Mexico basin, and their site fidelity has resulted in maternally related groups of females and young whales in this region.

Threats

Continued threats to sperm whale populations include vessel strikes, entanglement in fishing gear, competition for resources due to overfishing, pollution, loss of prey and habitat, and sound. NMFS' Recovery Plan for Sperm Whales (NMFS 2010b) identified four main categories of threats to the recovery of sperm whales in the Atlantic Ocean: (1) vessel interactions, (2) incidental capture in fishing gear, (3) habitat degradation, and (4) military operations. Loss of habitat can occur from multiple stressors including climate change, contaminant pollution and sound (Waring et al. 2016). Sound threats can include seismic surveys or propeller cavitation from large vessels, and this is heightened in areas of oil and gas activities or where shipping activity is high.

Vessels affect sperm whales via collisions and vessel sound. Sperm whales have been recorded spending periods of up to ten minutes "rafting" at the surface between deep dives (Watwood et al. 2006). This could make them exceptionally vulnerable to ship strikes. Studies on the behavior of sperm whales around whale watching boats suggest sperm whales change their diving and acoustic behavior in response to boats, but following frequent exposure, they become increasingly tolerant or habituated to the presence of vessels (Gordon et al. 1992; Markowitz et al. 2011).

Incidental entrapment and entanglement in fishing gear, especially gillnets set in deep water for pelagic fish (e.g., sharks, billfish, tuna), is of potential concern. In U.S. East coast waters, two incidents involving sperm whales and drift gillnets were reported between 1990 and 1995, both on Georges Bank. In 1990, a whale was found entangled and was released in "injured" condition. In 1995, another was found, also injured, and released while still carrying gear (Waring et al. 1997). Based on observer data, mortality of sperm whales from the drift gillnet fishery between 1989 and 1995 ranged from zero to 4.4 (CV 1.77) per year (Waring et al. 1997).

Two interactions between sperm whales and the pelagic longline fishery have been recorded in the Gulf of Mexico. Observer reports indicate that in 2008 an individual was entangled but released unharmed with no trailing gear. In 2015, a young individual was entangled and escaped with trailing gear attached, and therefore was categorized as a "serious injury."

Additionally, a stranded sperm whale has been documented with signs of human interaction (NOAA National Marine Mammal Health and Stranding database unpublished data 2002-2012). However, we do not have information on the source of the interaction.

The accumulation of stable pollutants (e.g. heavy metals, polycholorobiphenyls [PCBs], chlorinated pesticides [DDT, DDE, etc.], and polycyclic aromatic hydrocarbons [PAHs]) is of concern for sperm whales. The potential impact of coastal pollution may be an issue for this species in portions of its habitat, though little is known regarding the effect pollutants may have on individuals. Because sperm whales feed at high trophic levels and store the chemicals in their blubber, they are susceptible to chemical pollution. Sperm whales could potentially pass these chemicals to their offspring in their milk (Whitehead 2003). A population sensitivity analysis for the Gulf of Mexico sperm whales showed that if toxins, such as those found in oil spills, reduce the survivorship rate of the mature female sperm whales by as little as 2.2 percent, or the survivorship rate of mothers by 4.8 percent, the growth rate of the population would drop to a level that would result in a decline in the size of that population (Chiquet et al. 2013). The DWH oil spill and response impacted the Gulf of Mexico sperm whale population. The effects on sperm whales are described in greater detail in Section 4, as well as in the Final PDARP (found at http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan). Oil spills and response activities continue to threaten sperm whales.

Marine debris may be ingested by sperm whales as is the case with many marine animals. Debris entrained in the deep scattering layer where sperm whales feed could be mistaken for prey and incidentally ingested. Man-made sound and offshore energy development may also be adversely affecting habitat quality. Because of their apparent role as important predators of mesopelagic squid and fish, changing the abundance of sperm whales should affect the distribution and abundance of other marine species. Conversely, changes in the abundance of mesopelagic squid and fish from recently developed targeted fisheries could affect the distribution of sperm whales.

Sperm whales are potentially affected by military operations in a number of ways. Whales can be struck by vessels and disturbed by sonar and other anthropogenic sounds. Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins 1985; Watkins and Schevill 1975). They also stop vocalizing for brief periods when codas are being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

3.2.2 Sea Turtles

3.2.2.1 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.

3.2.2.2 Loggerhead Sea Turtle (NWA 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) NWA (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 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 United States 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 vertebrals, 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 United States, 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 1998).

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 29ºN; (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 2000); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS 2001).

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 zone[5]), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting 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 2001). 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 United States, 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).

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

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. 2008; 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 United States, 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. 2003; NMFS-SEFSC 2009; NMFS 2001; NMFS and USFWS 2008; TEWG 1998; TEWG 2000; 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.

<u>Peninsular Florida Recovery Unit</u>
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 3.4). 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 represented 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 non-significant 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. 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. However, it is important to note that with the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.) it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani, et al. 2019).

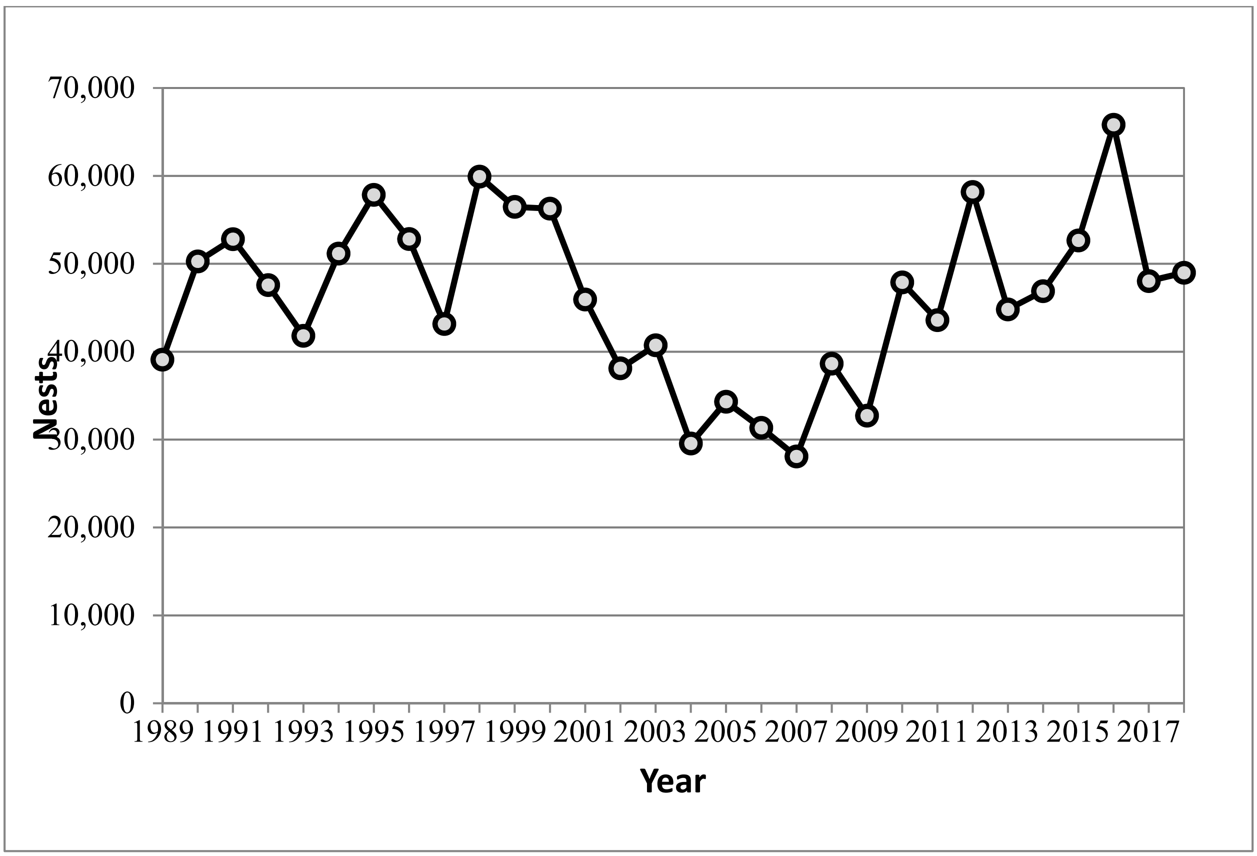

Figure 3.4 Loggerhead sea turtle nesting at Florida index beaches since 1989

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 3.4) 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, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016. Nesting in 2017 and 2018 declined relative to 2016, back to levels seen in 2013 to 2015, but then bounced back in 2019, breaking records for each of the three states and the overall Recovery Unit.

Table 3.4 Total Number of NRU Loggerhead Nests (GADNR, SCDNR, and NCWRC nesting datasets compiled at Seaturtle.org)

	Nests Recorded			
Year	**Georgia**	**South Carolina**	**North Carolina**	**Totals**
2008	1,649	4,500	841	**6,990**
2009	998	2,182	302	**3,472**
2010	1,760	3,141	856	**5,757**
2011	1,992	4,015	950	**6,957**
2012	2,241	4,615	1,074	**7,930**
2013	2,289	5,193	1,260	**8,742**
2014	1,196	2,083	542	**3,821**
2015	2,319	5,104	1,254	**8,677**
2016	3,265	6,443	1,612	**11,320**
2017	2,155	5,232	1,195	**8,582**
2018	1,735	2,762	765	**5,262**
2019	3,945	8,774	2,291	**15,010**

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 then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high but was still the second highest on record (Figure 3.5).

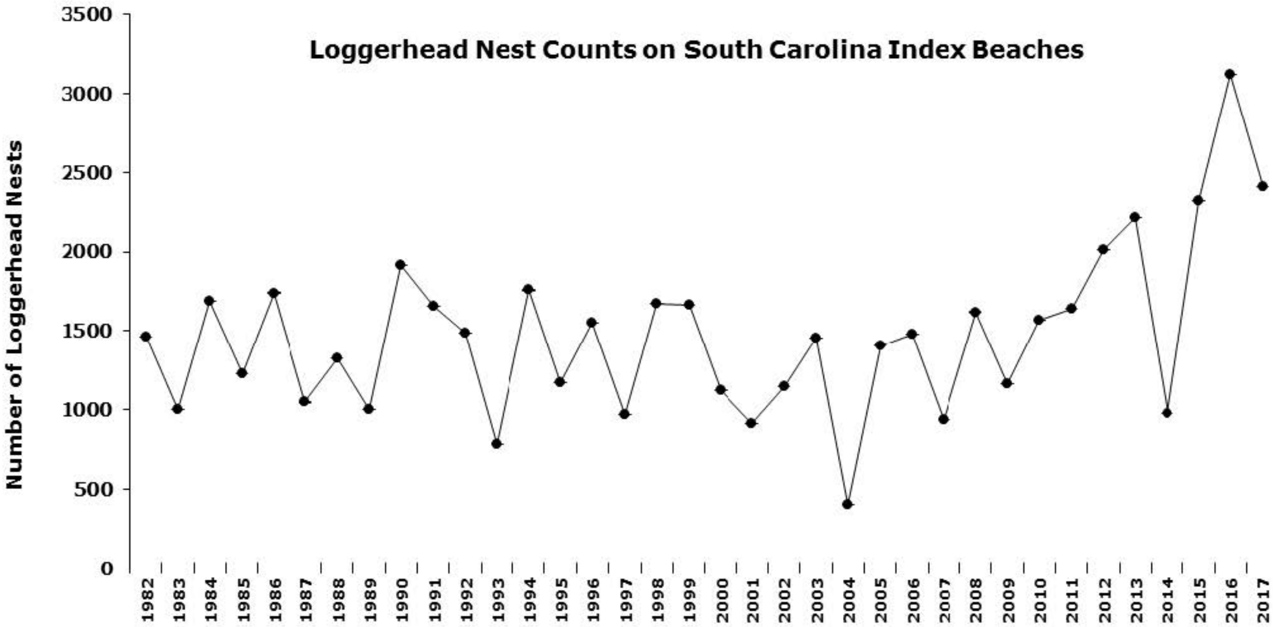

Figure 3.5 South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: http://www.dnr.sc.gov/seaturtle/nest.htm)

Other NWA 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 catch per unit effort

(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 United States 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 United States, 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 Southeast Fisheries Science Center 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) (NMFS-NEFSC 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 3.2.2.1. 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. 2008) 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. (2008) 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. 1991).

While oil spill impacts are discussed generally for all species in Section 3.2.2.1, specific impacts of the Deepwater Horizon (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 NWA 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 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 NGMRU), 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).

3.2.2.3 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 2000; 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 United States, 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 1989), 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 2.2-2.9 ± 2.4 in per year (5.5-7.5 ± 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. (2011) 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 increased to 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, indicated 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, followed by another decline to 11,090 in 2019 (Gladys Porter Zoo database 2019). 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 United States, 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). 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.

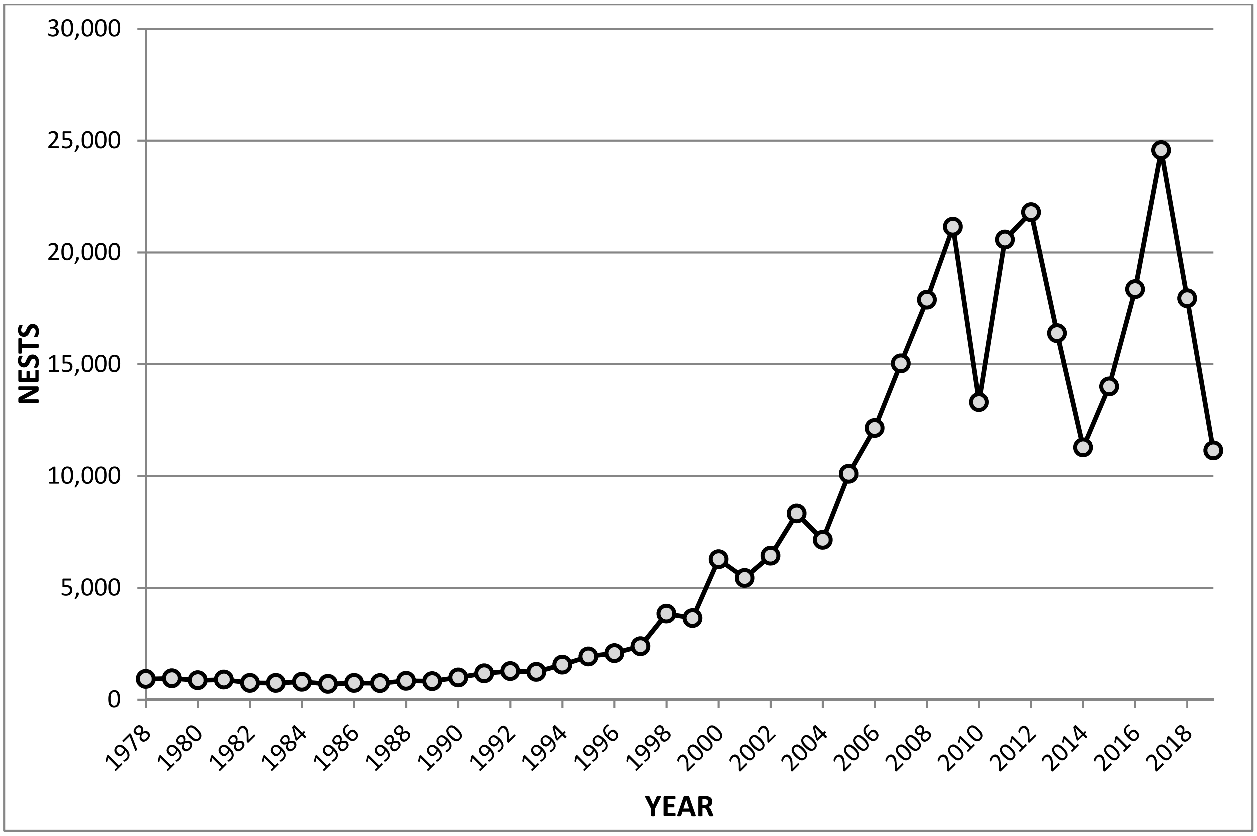

Figure 3.6 Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019)

Through modelling, Heppell et al. (2005) predicted the population is expected to increase at least 12-16% per year and could reach at least 10,000 females nesting on Mexico beaches by 2015. NMFS et al. (2011) 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 United States, and possibly other changes in vital rates (TEWG 1998; TEWG 2000). 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 3.2.2.1; 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 *arribadas*[6] 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, we have documented (via the Sea Turtle Stranding and Salvage Network data, https://www.fisheries.noaa.gov/national/marine-life-distress/sea-turtle-stranding-and-salvage-network) 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 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, 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

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

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 fisheries during the summer of 2012. During May-July of that year, observers reported 24 sea turtle interactions in the skimmer trawl fisheries. All but a single sea turtle were identified as Kemp's ridleys (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 fisheries. Due to this issue, a proposed 2012 rule to require TEDs in the skimmer trawl fisheries (77 FR 27411) was not implemented. Based on anecdotal information, these interactions were a relatively new issue for the inshore skimmer trawl fisheries. Following additional gear testing, however, we proposed a new rule in 2016 (81 FR 91097) to require TEDs with 3-inch (in) bar spacing for all vessels using skimmer trawls, pusher-head trawls, or wing nets. Ultimately, we published a final rule on December 20, 2019 (84 FR 70048), that requires all skimmer trawl vessels 40 feet and greater in length to use TEDs designed to exclude small sea turtles in their nets effective April 1, 2021. 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 3.2.2.1, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles 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, the calculation of unrealized nests and hatchlings was limited to Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011), 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).

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 estimated to have been 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, as up to 90,300 small juveniles Kemp's ridleys are estimated to have 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.

3.2.2.4 Green Sea Turtle (Information Relevant to All DPSs)

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 distinct population segments (DPSs) (81 FR 20057 2016) (Figure 3.7). 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 DPSs 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 United States.

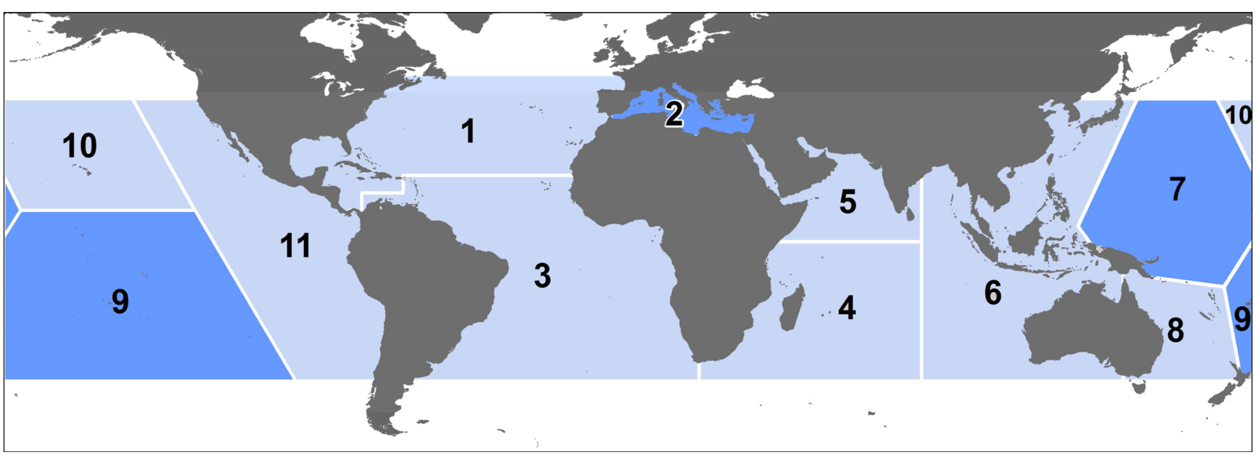

Figure 3.7 Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11.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

<u>North Atlantic DPS Distribution</u>

The NA DPS boundary is illustrated in Figure 1. 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 United States 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 United States 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 United States 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 1, 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 (applicable to all DPSs)
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 United States, 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 2007). 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 2007).

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).

Quintana Roo, Mexico, accounts for approximately 11% of nesting for the DPS (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 2007d). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpublished data, 2013, in 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 2007). 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 United States, green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida (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 (up to tens of nests) (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 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 resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years.

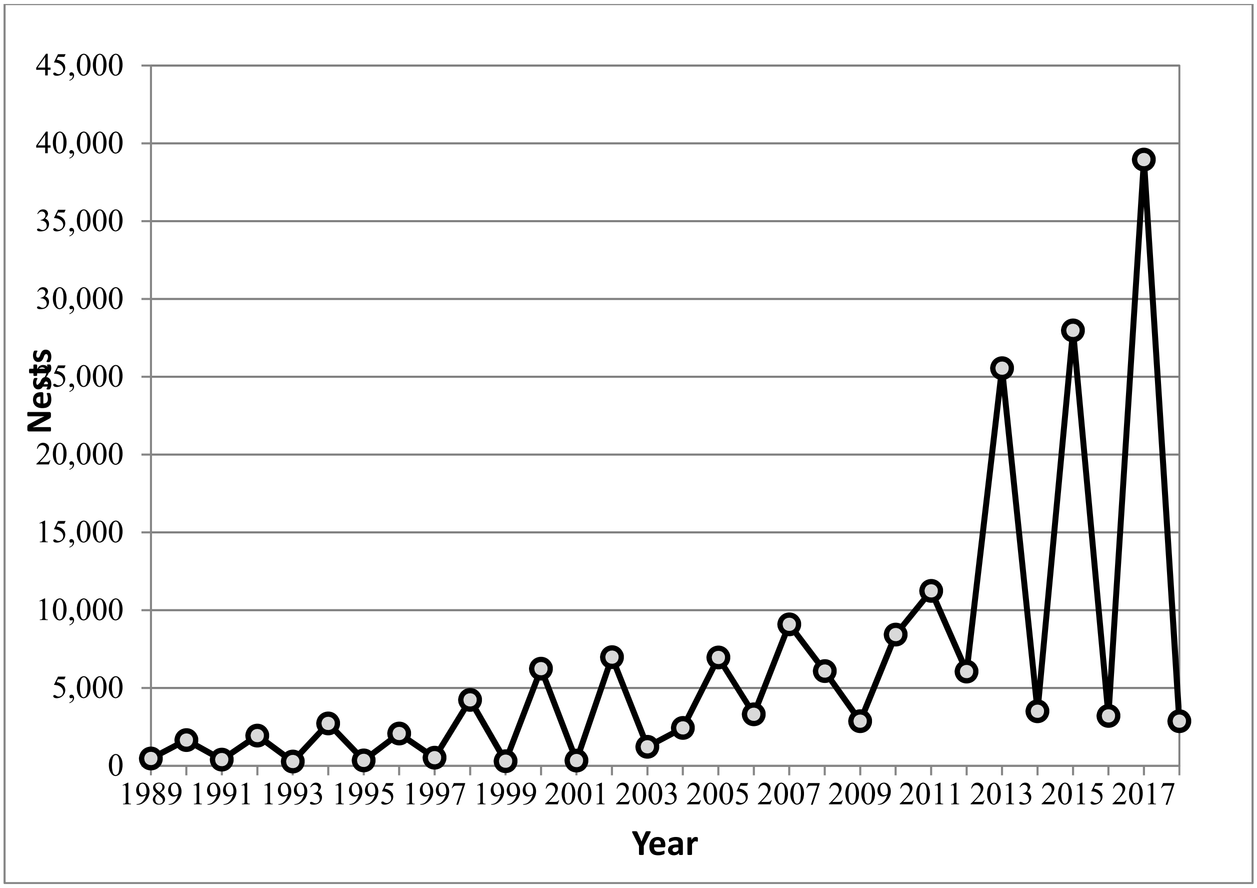

Figure 3.8. Green sea turtle nesting at Florida index beaches since 1989

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 percent 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 United States, 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 3.2.2.1.

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 United States 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 3.2.2.1, 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 juveniles 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 Deepwater Horizon oil spill of 2010 (DWH), 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).

3.2.2.5 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 United States, 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 1998; 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 United States occurs in Puerto Rico and the U.S. Virgin Islands, specifically on Mona Island and BIRNM, respectively. Although nesting within the continental United States 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 2007).

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) (Boulan 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 2007). 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 United States, 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. The conservation measures implemented when BIRNM was expanded in 2001 most likely explains this increase.

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 2007).

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 3.2.2.1. 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 3.2.2.1, 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.

3.2.2.6 Leatherback Sea Turtle

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 1998). 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),[7] a thick layer of insulating fat (Davenport et al. 1990; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),[8] 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 2001).

While leatherbacks will look for food in coastal waters, they appear to prefer the open ocean at all life stages (Heppell et al. 2003). 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 are jellies (e.g., medusae, siphonophores, and salps), which commonly occur in temperate and northern or sub-arctic latitudes and likely has a strong

[7] 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. As the warm blood flows away from the heart, it passes much of its heat to the colder blood returning to the heart via the veins. This conserves heat by recirculating it back to the body's core.
[8] "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.

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.

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. 2003; 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 United States), 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 United States, 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 lengths 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 had been less clear than the Pacific population, which has shown dramatic declines at many nesting sites (Spotila et al. 2000; Santidrián Tomillo et al. 2007; Sarti Martínez et al. 2007). 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 Turtle Expert Working Group 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 (2020).

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 2001). 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,[9] while the number of nests was declining at beaches in Guiana (Hilterman et al. 2003). 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 at 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). Tiwari et al. (2013) report an estimated three-generation abundance change of -72%, -24%, and +6% for

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

Tortuguero, Gandoca, and Pacuare, respectively. Further decline of almost 6% annual geometric mean from 2008-2017 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). Tiwari et al. (2013) 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, Tiwari et al. (2013) 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 3.9 and Table 3.5). A similar pattern was also observed statewide (Table 3.5). 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 showed growth on Florida's east coast beaches. Tiwari et al. (2013) 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).

Table 3.5 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

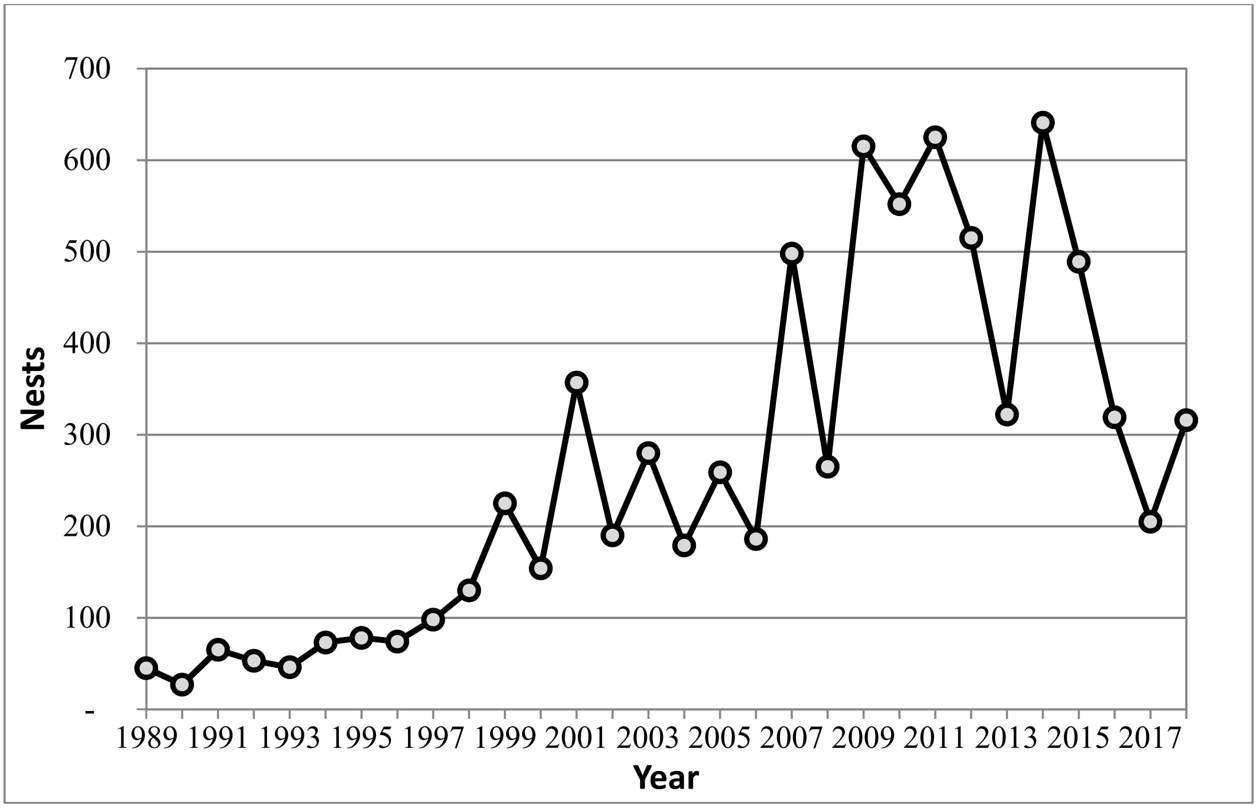

Figure 3.9 Leatherback sea turtle nesting at Florida index beaches since 1989

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, TEWG (2007) determined that between 1988 and 2003, there was a positive annual average growth rate between 1.07% and 1.08% for the Brazilian stock. TEWG (2007) estimated an annual average growth rate between 1.04% and 1.06% for the South African stock.

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 USFWS (2013) suggested 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 3.2.2.1; 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. This 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 a plastic bags (Mrosovsky et al. 2009). Balazs (1985) 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 3.2.2.1, 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 2007). Several studies have shown leatherback distribution is influenced by jellyfish abundance ((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 3.2.2.1, 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 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, it was concluded that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. 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.

3.2.2.7 Olive Ridley Sea Turtle

The olive ridley sea turtle was listed on July 28, 1978, with all breeding populations listed as threatened except for the Pacific coast of Mexico population, which is endangered. There have also been recommendations that the western Atlantic olive ridley populations be reclassified as endangered (Reichart 1993). The olive ridley is a small, hard-shelled sea turtle with an olive colored shell. It typically occurs within the tropical regions of the Pacific, Atlantic, and Indian Oceans. This species does not nest in the United States, but during feeding migrations, olive ridley turtles nesting in the Pacific may disperse into waters of the southwestern U.S., occasionally as far north as Oregon. Olive ridleys are more abundant and widespread in the Pacific than in the Atlantic, especially the western Atlantic where olive ridley populations are limited in size and range. There is no Recovery Plan for Atlantic olive ridleys unlike the Pacific populations, as Atlantic olive ridleys are considered outside of U.S. jurisdiction except in very rare cases.

The olive ridley is most noted for its massive nesting aggregations, known as arribadas, with thousands of females nesting in large simultaneous waves over small stretches of beach. Arribadas may be precipitated by climatic events, such as a strong offshore wind, or by certain phases of the moon and tide; however, there is a major element of unpredictability regarding the trigger and timing at all arribada sites. Although not every adult female participates in these arribadas, the vast majority do.

Pacific Ocean
In the eastern Pacific, olive ridleys nest primarily on beaches from Mexico south to at least Colombia (NMFS and USFWS 1995) with major nesting beaches at Escobilla, Mexico; La Flor, Nicaragua; and Ostional and Nancite, Costa Rica. Declines in nesting have been documented for Playa Nancite, Costa Rica. However, other nesting populations along the Pacific coast of Mexico and Costa Rica appear stable or increasing (NMFS 2004). When not at the nesting areas, adult olive ridleys are generally found in warm waters from Baja California, Mexico to Chile (Silva-Batiz et al. 1996). In the western Pacific, nesting information is not available for several countries, but information from Indonesia suggests an increase in nesting, while information from Malaysia and Thailand suggests that nesting has declined to very low levels in those countries (NMFS 2004). In the Indian Ocean, olive ridleys nest in great abundance in eastern India and Sri Lanka, although minor nesting also occurs at other localities. Gahirmatha, located in the Bhitarkanika Wildlife Sanctuary, India, supports perhaps one of the largest nesting populations in the world with an average of 398,000 females nesting in a given year. These populations, however, are suffering high mortality from nearshore gillnets and trawl fisheries.

Atlantic Ocean
Western Atlantic *arribada* nesting populations are currently very small. Recent data indicate the Suriname/French Guiana nesting population may still be threatened by incidental capture in the shrimp trawl fishery. Nesting data from French Guiana/Suriname during the 2002-2006 nesting seasons indicate that while nesting in Suriname continues at very low levels, nesting in French Guiana and overall nesting appears comparable to levels recorded for both countries about two decades ago. This may indicate a shift from nesting beaches in Suriname to French Guiana and reflect the dynamic aspects of beach erosion and accretion in the region. The other nesting population in Brazil, for which no long term data are available, is small, but increasing. In the eastern Atlantic, long-term empirical data are not available and thus the abundance and trends of this population cannot be assessed at this time (NMFS and USFWS 2014). As is the case with olive ridleys in the Pacific, the overall range of the species is much broader than the nesting range. Sporadic sightings of olive ridleys have occurred in the Caribbean and recently in Florida, and a confirmed individual was captured in 2003 during the experimental longline fishery in the NED, in the northern Atlantic Ocean. However, those areas are not thought to be part of the species' normal range. Other documented interactions with olive ridley sea turtles and the U.S. HMS PLL fishery have all occurred in the TUN fishery reporting area, off northern South America, which is within the normal range of the Atlantic olive ridley.

Life history and Atlantic Distribution
Age at sexual maturity for olive ridleys is not known, but if similar to its close relative the Kemp's ridley, it would be 7 to 15 years. Olive ridleys typically nest 1 to 3 times per season, producing about 100 to110 eggs on each occasion. The inter-nesting interval is variable, but for most localities it is approximately 14 days for solitary nesters and 28 days for arribada nesters. Incubation takes about 50 to 60 days.

In the western Atlantic, olive ridleys have been reported at sea as far north as the Grand Banks Region and as far south as Uruguay, encompassing a range between 43°N and 34°S (Foley et al. 2003; Fretey 1999; Stokes and Epperly 2006). However, they are most common in the waters of Guyana, Suriname, French Guiana, and Brazil and are not common elsewhere in the region. Female olive ridleys appear to remain in neritic waters during (Plot et al. 2012) and after breeding (Pritchard 1976; Reichart 1993). They forage on the continental plateau of Suriname and Guyana (Feuillet and de Thoisy 2007 as cited in de Boer 2013; Georges et al. 2008). There is little geographic overlap between the olive ridleys nesting in French Guiana/Suriname and those from Brazil (Godfrey and Chevalier 2004). Historic tag returns from females that nested in French Guiana/Suriname indicate that turtles migrate either south to foraging areas ranging from eastern Guyana to Amapa (Brazil), or north, to foraging areas ranging from the mouth of the Orinoco River to the islands of Trinidad and Tobago, and Margarita (Pritchard 1973; Schulz 1975). Tag returns from females that nested in Sergipe have been recovered in Sergipe or farther south in Brazil (Marcovaldi et al. 2000).

Information on olive ridleys in the eastern Atlantic is limited, but it is clear that olive ridleys are common throughout this region (Fretey et al. 2005). The species has been confirmed, or is thought to occur, along the coast between Mauritania and South Africa. The highest densities have been recorded in the Gulf of Guinea between the Ivory Coast and Gabon. Similar to the western Atlantic, there are few pelagic records of olive ridleys from the eastern Atlantic Ocean. In the region, reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds located near nesting beaches (Cornelius 1986; Hughes and Richard 1974; Kalb et al. 1995; Maxwell et al. 2011; Plotkin et al. 1991, 1996, 1997; Pritchard 1969). A significant proportion of the breeding also takes place far from shore (Pitman 1991, Kopitsky et al. 2000), and it is possible that some males and females may not migrate to nearshore breeding aggregations at all. Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females en route to nearshore breeding grounds and nesting beaches (Plotkin 1994; Plotkin, et al. 1994, 1996; Kopitsky et al. 2000). During the internesting interval, females stayed in shallow waters (less than 50 m depth) within 30 km of the nesting beach in Gabon (Maxwell et al. 2011). Post-nesting females from Gabon and Angola travelled a minimum straight-line distance between 694 and 9,182 km within oceanic waters and largely in a southerly direction (Pikesley et al. 2013).

As described above, there are no known nesting sites for olive ridleys in U.S. waters. In the past several years, olive ridley turtles have been occasionally documented in stranding records in the southeastern U.S. and U.S. Caribbean, where they had never been documented before. In addition, the documented capture of an olive ridley in the NED experiment in 2003 is the first known interaction with the HMS PLL fishery. Caution should be used to avoid over interpreting these very few occurrences, but the change from absence to presence in U.S. Atlantic records is notable. While still very rare, subsequent documented interactions have occurred with the HMS PLL fishery, but all within the TUN fishery reporting area off northern South America, which is within the normal range for the species. It is also notable that unlike most of the Pacific populations, which exhibit more widespread movements and frequent use of the oceanic environment even as adults, western Atlantic Olive Ridleys appear to remain in neritic waters after breeding (Pritchard 1976, Reichart 1993).

There are surprisingly few data relating to the feeding habits of the olive ridley. However, those reports that do exist suggest that the diet in the western Atlantic and eastern Pacific includes crabs, shrimp, rock lobsters, jellyfish, and tunicates. In some parts of the world, it has been reported that the principal food is algae.

Population Dynamics and Status
The olive ridley is widely regarded as the most abundant sea turtle in the world because of the continued existence of several large arribadas. However, since its listing under the ESA, there has been a decline in abundance of this species in the western Atlantic, probably the result of continued direct and incidental take, particularly in shrimp trawl nets and nearshore gill nets. The western North Atlantic (Suriname, French Guiana, and Guyana) nesting population has declined more than 80 percent since 1967. However, as noted above nesting in Suriname and French Guiana may be showing signs of stabilizing, and the very small nesting population in Brazil may be increasing. Similar declines have been seen in the Pacific, although some nesting populations appear to be stable or increasing as described above. The Indian Ocean continues to support one of the largest nesting populations in the world. However, these populations are also known to suffer high anthropogenic mortality from fishery interactions.

Threats
The decline of this species is primarily due to human activities, including the direct harvest of adults and eggs, incidental capture in commercial fisheries, and loss of nesting habitat. However, their characteristic form of nesting, the arribada, also leaves then susceptible to natural predation (as well as poaching) and a high incidence of incidental nest destruction by other nesting turtles. Even the close proximity of a rotting nest can lead to bacterial contamination and destruction of all or part of the surrounding nests (NMFS 2004).

Summary of Status for Olive Ridley Sea Turtles
The western North Atlantic (Suriname, French Guiana, and Guyana) nesting population has declined more than 80 percent since 1967 but may be stabilizing. Anthropogenic impacts similar to those experienced by other sea turtle species (i.e., such as fishing interactions and poaching) appear to be primarily responsible for the decline. There are no olive ridley turtle nesting sites within the U.S. In the past several years, however, olive ridley turtles have been occasionally documented in stranding records in the southeastern U.S. and U.S. Caribbean, where they had never been documented before. In addition, in 2003, the NED experimental longline fishery in the northern western Atlantic documented capture of an olive ridley sea turtle. Caution should be used to avoid over interpreting these very few occurrences, but the change from absence to presence in U.S. Atlantic records is notable.

3.2.3 Giant Manta Ray

NMFS listed the giant manta ray (*Manta birostris*) as threatened under the ESA effective February 21, 2018 (83 FR 2916). NMFS determined that the designation of critical habitat is not prudent on December 5, 2019 (84 FR 66652). On December 4, 2019, NMFS published a recovery outline for the giant manta ray (NMFS 2019). The recovery outline serves as an interim guidance to direct recovery efforts for giant manta ray.

Species Description and Distribution
The giant manta ray is the largest living ray, with a wingspan reaching a width of up to 7 m (23 ft), and an average size between 4-5 m (15-16.5 ft). The giant manta ray is recognized by its large diamond-shaped body with elongated wing-like pectoral fins, ventrally placed gill slits, laterally placed eyes, and wide terminal mouth. In front of the mouth, it has two structures called cephalic lobes that extend and help to introduce water into the mouth for feeding activities (making them the only vertebrate animals with three paired appendages). 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). There are bright white shoulder markings on the dorsal side that form two mirror image right-angle triangles, creating a T-shape on the upper shoulders.

The giant manta ray can be found in all ocean basins. In terms of range, within the Northern hemisphere, the species has been documented as far north as southern California and New Jersey on the United States west and east coasts, respectively, and Mutsu Bay, Aomori, Japan, the Sinai Peninsula and Arabian Sea, Egypt, and the Azores Islands (Gudger 1922; Kashiwagi et al. 2010; Moore 2012; CITES 2013). In the Southern Hemisphere, the species occurs as far south as Peru, Uruguay, South Africa, New Zealand and French Polynesia (Mourier 2012; CITES 2013). Within its range, the giant manta ray inhabits tropical, subtropical, and temperate bodies of water and is commonly found offshore, in oceanic waters, and near productive coastlines (Figure 3.10) (Marshall et al. 2009; Kashiwagi et al. 2011).

Figure 3.10 The Extent of Occurrence (light blue) and Area of Occupancy (dark blue) based on species distribution. Source Lawson et al. 2017.

Life History Information
Giant manta rays make seasonal long-distance migrations, aggregate in certain areas and remain resident, or aggregate seasonally (Dewar et al. 2008; Graham et al. 2012; Girondot et al. 2015; Stewart et al. 2016). High rates of site residency and fidelity have been demonstrated, particularly in manta rays (Jaine et al., 2014; Stewart et al., 2016a; Couturier et al., 2018), with examples of seasonal migration between known aggregation sites up to 750 km apart (Couturier

et al., 2011; Germanov and Marshall, 2014). The giant manta ray is a seasonal visitor along productive coastlines with regular upwelling, in oceanic island groups, and at 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; J. Pate, Florida Manta Project, unpublished data).

Giant manta rays are known to aggregate in various locations around the world in groups usually ranging from 100-1,000 (Notarbartolo-di-Sciara and Hillyer 1989; Graham et al. 2012; Venables 2013). These sites function as feeding sites, cleaning stations, or sites where courtship interactions take place (Heinrichs et al. 2011; Graham et al. 2012; Venables 2013). The appearance of giant manta rays in these locations is generally predictable. For example, food availability due to high productivity events tends to play a significant role in feeding site aggregations (Notarbartolo-di-Sciara and Hillyer 1989; Heinrichs et al. 2011; Jaine et al. 2012). Giant manta rays have also been shown to return to a preferred site of feeding or cleaning over extended periods of time (Dewar et al. 2008; Graham et al. 2012; Medeiros et al. 2015). In addition, giant and reef manta rays in Keauhou and Ho"ona Bays in Hawaii, appear to exhibit learned behavior. These manta rays learned to associate artificial lighting with high plankton concertation (primary food source) and shifted foraging strategies to include sites that had artificially lighting at night (Clark 2010). While little is known about giant manta ray aggregation sites, the Flower Garden Banks National Marine Sanctuary (FGBNMS) and the surrounding region might represent the first documented nursery habitat for giant manta ray (Stewart et al. 2018). Stewart et al. (2018) found that the FGBNMS provides nursery habitat for juvenile giant manta rays because small age classes have been observed consistently across years at both the population and individual level. The FGBNMS may be an optimal nursery ground because of its location near the edge of the continental shelf and proximity to abundant pelagic food resources. In addition, small juveniles are frequently observed along a portion of Florida's east coast, indicating that this area may also function as a nursery ground for juvenile giant manta rays. Since directed visual surveys began in 2016, juvenile giant manta rays are regularly observed in the shallow waters (less than 5 m depth) from Jupiter Inlet to Boynton Beach Inlet (J Pate, Florida Manta Project, unpublished data). However, the extent of this purported nursery ground is unknown as the survey area is limited to a relatively narrow geographic area along Florida's east coast.

The giant manta ray appears to exhibit a high degree of plasticity in terms of its use of depths within its habitat. Tagging studies have shown that the giant manta rays conduct night descents from 200-450m depths (Rubin et al. 2008; Stewart et al. 2016) and are capable of diving to depths exceeding 1,000 m (A. Marshall et al. unpubl. data 2011, cited in Marshall et al. (2011). 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. 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).

Giant manta rays primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderately sized fishes (Miller and Klimovich 2017). 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. When feeding, giant manta rays hold their cephalic lobes 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 repeatedly) and creating feeding chains with other mantas to maximize prey intake.

The giant manta ray is viviparous (i.e., gives birth to live young). They are slow to mature and have very low fecundity and typically give birth to only one pup every two to three years. Gestation lasts approximately 10-14 months. Females are only able to produce between 5 and 15 pups in a lifetime (CITES 2013; Miller and Klimovich 2017). The giant manta ray has one of the lowest maximum population growth rates of all elasmobranchs (Dulvy et al. 2014; Miller and Klimovich 2017). The giant manta ray's 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 (Miller and Klimovich 2017). Males are estimated to mature at around 3.8 m disc width (slightly smaller than females) and females at 4.5 m disc width (Rambahiniarison et al. 2018).

Status and Population Dynamics

There are no current or historical estimates of global abundance of giant manta rays, with most estimates of subpopulations based on anecdotal observations. The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES, 2013) found that only ten populations of giant manta rays had been actively studied, 25 other aggregations have been anecdotally identified, all other sightings are rare, and the total global population may be small. Subpopulation abundance estimates range between 42 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. Ecuador is thought to be home to the largest identified population (n=1,500) of giant manta rays in the world, with large aggregation sites within the waters of the Machalilla National Park and the Galapagos Marine Reserve (Hearn et al. 2014). Within the Indian Ocean, numbers of giant manta rays identified through citizen science in Thailand's waters (primarily on the west coast, off Khao Lak and Koh Lanta) was 288 in 2016. These numbers reportedly surpass the estimate of identified giant mantas in Mozambique (n=254), possibly indicating that Thailand may be home to the largest aggregation of giant manta rays within the Indian Ocean (MantaMatcher 2016). 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, although similar declines have not been observed in areas with other subpopulations, such as Mozambique and Ecuador. In the Atlantic, populations are likely small and sparsely distributed. In the U.S. Atlantic, the giant manta ray appears to have a seasonal pattern of occurrence along Florida's east coast, showing up with greater frequencies (and in greater numbers) in the spring and summer months. Available sightings data indicates that manta rays are conduct seasonal visits to Florida's inshore waters. The numbers, location, and peak timing of the manta rays to this area varies by year along northeast Florida (H. Webb unpublished data). In 2015, aerial survey conducted by the Georgia Aquarium peaked at 1,144 manta ray sighted in the inshore waters of northeast Florida, but with notable decline in manta rays observed in the study area since 2015 (H. Webb unpublished data). Juvenile manta rays are also regularly observed inshore off southeast Florida. Since 2016, researchers with the Marine Megafauna Foundation have been conducting annual surveys along a small transect off Palm Beach, Florida, between Jupiter Inlet and Boynton Beach Inlet (~44 km, 24 nautical miles) (J. Pate, MMF, pers. comm. to M. Miller, NMFS OPR, 2018). Survey results indicate juvenile manta rays are present in this area most of the year (observations span from May to December). The re-sightings also suggest a portion of those manta rays remain in the area for extended periods of time or return in subsequent years (J. Pate unpublished data). In the Gulf of Mexico, within the Flower Garden Banks National Marine Sanctuary, 95 unique individuals have been recorded between 1982 and 2017 (Stewart et al., 2018).

Threats

The giant manta ray faces many threats, including fisheries interactions, environmental contaminants (microplastics, marine debris, petroleum products, etc.), vessel strikes, entanglement, and global climate change. Overall, the predictable nature of their appearances, combined with slow swimming speed, large size, and lack of fear towards humans, may increase their vulnerability to threats (O'Malley et al. 2013; CMS 2014). The ESA status review determined that the greatest threat to the species results from fisheries related mortality (Miller and Klimovich 2017; 83 FR 2916, January 22, 2018).

Commercial Harvest and Fisheries Bycatch

Commercial harvest and incidental bycatch in fisheries is cited as the primary cause for the decline in the giant manta ray and threat to future recovery (Miller and Klimovich 2017). We anticipate that these threats will continue to affect the rate of recovery of the giant manta ray. Worldwide giant manta ray catches have been recorded in at least 30 large and small-scale fisheries covering 25 countries (Lawson et al. 2016). Demand for the gills of giant manta rays and other mobula rays has risen dramatically in Asian markets. With this expansion of the international gill raker market and increasing demand for manta ray products, estimated harvest of giant manta rays, particularly in many portions of the Indo-Pacific, frequently exceeds numbers of identified individuals in those areas and are accompanied by observed declines in sightings and landings of the species of up to 95% (Miller and Klimovich 2017). In the Indian Ocean, manta rays (primarily giant manta rays) are mainly caught as bycatch in purse seine and gillnet fisheries (Oliver et al. 2015). In the western Indian Ocean, data from the pelagic tuna purse seine fishery suggests that giant manta and mobula rays, together, are an insignificant portion of the bycatch, comprising less than 1% of the total non-tuna bycatch per year (Romanov 2002; Amandè et al. 2008). In the U.S., bycatch of giant manta rays has been recorded in the coastal migratory pelagic gillnet, gulf reef fish bottom longline, Atlantic shark gillnet, pelagic bottom longline, and trawl fisheries. Incidental capture of giant manta ray is also a rare occurrence within U.S. Atlantic and Gulf of Mexico, with the majority that are caught released alive.

The giant manta ray is also incidentally captured by recreational fishers using vertical line (i.e., handline, bandit gear, and rod-and-reel). Researchers report giant manta rays having evidence of recreational gear interactions along the east coast of Florida (i.e., manta rays have embedded fishing hooks with attached trailing monofilament line) (J. Pate, Florida Manta Project, unpublished data). There are also anecdotal reports of recreational interactions with giant manta rays. For example, recreational fishers will search for giant manta rays while targeting cobia, as cobia often accompany giant manta rays (anglers will cast at manta rays in an effort to hook cobia). Giant manta rays are commonly observed swimming near or underneath public fishing piers where they may become foul-hooked. The current threat of mortality associated with recreational fisheries generally is expected to be low, given that we have no reports of recreational fishers retaining giant manta ray. However, bycatch in recreational fisheries generally remains a potential threat to the species.

Vessel Strike

Vessel strikes can injure or kill giant manta rays, decreasing fitness or contributing to non-natural mortality (Deakos et al. 2011; Couturier et al. 2012). Giant manta rays do not surface to breathe, but they can spend considerable time in surface waters, while basking and feeding, where they are more susceptible to vessel strikes (McGregor et al., 2019). Manta rays also show little fear of vessels which increases the risk of vessel strikes (Deakos 2010; C. Horn personal observation). Along Florida's southeast coast, five giant manta rays have been struck by vessels from 2016 - 2018; individuals had injuries (i.e., fresh or healed dorsal surface propeller scars) consistent with a vessel strike. These interactions were observed by researchers conducting surveys from Boynton Beach to Jupiter, Florida (J. Pate, Florida Manta Project, unpublished data). The giant manta ray is frequently observed in nearshore coastal waters and feeding within and around inlets. As vessel traffic is concentrated in and around inlets and nearshore waters, this overlap exposes the giant manta ray in these locations to an increased likelihood of potential vessel strike. Yet, few instances of confirmed or suspected mortalities of giant manta ray attributed to vessel strike injury (e.g., via strandings) have been documented. This lack of documented mortalities could also be the result of other factors that influence carcass detection (i.e., wind, currents, scavenging, decomposition etc.). In addition, manta rays appear to be able to heal from wounds very quickly, while high wound healing capacity is likely to be beneficial for their long-term survival, the fitness cost of injuries and number vessel strikes occurring may be masked (McGregory et al., 2019).

Microplastics

The rising level of plastic debris in our oceans is a large-scale environmental problem with wide ranging impacts (van Sebille et al., 2015; Worm et al., 2017; Germanov et al., 2018). While large debris also impacts marine organisms (e.g., turtles, birds, sharks, and mammals), once broken down to microscopic sizes through environmental exposure, microplastics are of growing concern as they easily enter food webs (Andrady, 2011;Worm et al., 2017). Microplastics, generally referred to as plastics < 5 mm in diameter (Andrady, 2011), are comparable in size to, or smaller than zooplankton, an integral component in marine ecosystems and the primary food for planktivores (Germanov et al., 2019). Filter-feeding megafauna are particularly susceptible to high levels of microplastic ingestion and exposure to associated toxins due to their feeding strategies, target prey, and, for most, habitat overlap with microplastic pollution hotspots (Germanov et al. 2018). Giant manta rays are filter feeders, and, therefore can ingest microplastics directly from polluted water or indirectly through-contaminated planktonic prey (Miller and Klimovich 2017). The effects of ingesting indigestible particles include blocking adequate nutrient absorption and causing mechanical damage to the digestive tract. Microplastics can also harbor high levels of toxins and persistent organic pollutants, and introduce these toxins to organisms via ingestion. These toxins can bioaccumulate over decades in long-lived filter feeders, leading to a disruption of biological processes (e.g., endocrine disruption), and potentially altering reproductive fitness (Germanov et al. 2018). Jambeck et al. (2015) found that the Western and Indo- Pacific regions are responsible for the majority of plastic waste. These areas also happen to overlap with some of the largest known aggregations of giant manta rays. For example, in Thailand, where recent sightings data have identified over 288 giant manta rays (MantaMatcher 2016), mismanaged plastic waste is estimated to be on the order of 1.03 million tonnes annually, with up to 40% of this entering the marine environment (Jambeck et al. 2015). Approximately 1.6 million tonnes of mismanaged plastic waste is being disposed of in Sri Lanka, again with up to 40% entering the marine environment (Jambeck et al. 2015), potentially polluting the habitat used by the nearby Maldives aggregation of manta rays. A recent evaluation of plastic abundance within manta ray feeding grounds in Indonesia found extremely high ingestion rates of plastic for manta rays. Germanov et al (2019) found that during peak season reef manta rays were estimated to be ingesting up to 980 g of plastic per kg of plankton (Germanov et al., 2019). As filter-feeders are likely to feed from several locations, and plastic abundance is likely to vary along these planes, quantifying the exposure of filter-feeders to pollutants can serve as a proxy for plastic exposure (Fossi et al., 2014, 2017) and might better capture the level of risk to individuals and populations (Germanov et al. 2019).

Mooring and Anchor Lines

Mooring and boat anchor line entanglement may also wound giant manta rays or cause them to drown (Deakos et al. 2011; Heinrichs et al. 2011). There are numerous anecdotal reports of giant manta rays becoming entangled in mooring and anchor lines (C. Horn, NMFS, unpublished data), as well as documented interactions encountered by other species of manta rays (C. Horn, NMFS, unpublished data). For example, although a rare occurrence, reef manta rays on occasion entangle themselves in anchor and mooring lines. Deakos (2010) suggested that manta rays become entangled when the line makes contact with the front of the head between the cephalic lobes, the animal's reflex response is to close the cephalic lobes, thereby trapping the rope between the cephalic lobes, entangling the manta ray as the animal begins to roll in an attempt to free itself. In Hawaii, on at least two occasions, a reef manta ray was reported to have died after entangling in a mooring line (A. Cummins, pers. comm. 2007, K. Osada, pers. comm. 2009; cited in Deakos 2011). In Maui, Hawaii, Deakos et al. (2011) observed that 1 out of 10 reef manta rays had an amputated or disfigured non-functioning cephalic lobe, likely a result of line entanglement. Mobulid researchers indicate that entanglements may significantly affect the manta rays fitness (Deakos et al. 2011; Heinrichs et al. 2011; Couturier et al. 2012; CMS 2014; Germanov and Marshall 2014; Braun et al. 2015). 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 species.

Climate Change Effects
Because giant manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the impacts of climate change compared to other sharks and rays (Chin et al. 2010). However, as giant manta rays frequently rely on coral reef habitat for important life history functions (e.g., feeding, cleaning) and depend on planktonic food resources for nourishment, both of which are highly sensitive to environmental changes (Brainard et al. 2011; Guinder and Molinero 2013), climate change is likely to have an impact on the distribution and behavior. Coral reef degradation from anthropogenic causes, particularly climate change, is projected to increase through the future. Specifically, annual, globally averaged surface ocean temperatures are projected to increase by approximately 0.7 °C by 2030 and 1.4 °C by 2060 compared to the 1986-2005 average (IPCC 2013), with the latest climate models predicting annual coral bleaching for almost all reefs by 2050 (Heron et al. 2016). Declines in coral cover have been shown to result in changes in coral reef fish communities (Jones et al. 2004; Graham et al. 2008). Therefore, the projected increase in coral habitat degradation may potentially lead to a decrease in the abundance of fish that clean giant manta rays (e.g., *Labroides* spp., *Thalassoma* spp., and *Chaetodon* spp.) and an overall reduction in the number of cleaning stations available to manta rays within these habitats. Decreased access to cleaning stations may negatively affect the fitness of giant manta rays by hindering their ability to reduce parasitic loads and dead tissue, which could lead to increases in diseases and declines in reproductive fitness and survival rates.

Changes in climate and oceanographic conditions, such as acidification, are also known to affect zooplankton structure (size, composition, and diversity), phenology, and distribution (Guinder and Molinero 2013). As such, the migration paths and locations of both resident and seasonal aggregations of giant manta rays, which depend on these animals for food, may similarly be altered (Couturier et al. 2012). As research to understand the exact impacts of climate change on marine phytoplankton and zooplankton communities is still ongoing, the severity of this threat has yet to be fully determined (Miller and Klomovich, 2017).

3.2.4 Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS

Four of six 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 3.34). 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.

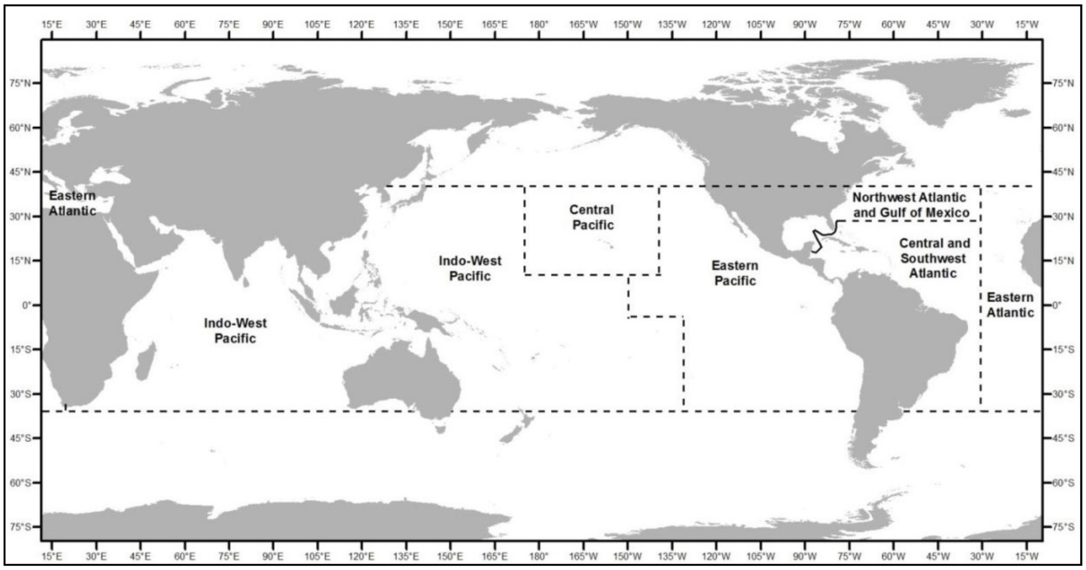

Figure 3.11 Scalloped hammerhead shark DPS boundaries (Source: 78 FR 20717; April 5, 2013).
Note: The Northwest Atlantic/Gulf of Mexico and Central Pacific DPSs are not listed under the ESA.

Species Description and Distribution
All hammerhead sharks belong to the family *Sphyrnidae* and are classified as requiem sharks (order Carcharhiniformes). 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 is found throughout the world and lives in coastal warm temperate and tropical seas. It 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). In the western Atlantic Ocean, the scalloped hammerhead's range extends from the northeast coast of the U.S. (New Jersey) to Florida and on to Brazil, including the Gulf of Mexico and Caribbean Sea.

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 (Duncan and Holland 2006; Hearn et al. 2010; Holland et al. 1993) (Bessudo et al. 2011). 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 mi [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, especially near the Galapagos, Malpelo, Cocos and Revillagigedo Islands, and within the Gulf of California (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 Informati*on
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 both the Northwest Atlantic and Gulf of Mexico DPS, and the Central and Southwest Atlantic DPS) of scalloped hammerheads has experienced severe declines over the past few decades. 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 HMS PLL 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).

For 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).

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. Miller et al. (2014) concluded that though abundance numbers for the Central and Southwest Atlantic DPS are unavailable, they are likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS. It is likely that scalloped hammerheads in the Central and Southwest Atlantic DPS have experienced at least the same level of decline as observed in the Northwest Atlantic and Gulf of Mexico DPS since the early 1980s (i.e., 83%). However, unlike the Northwest Atlantic and Gulf of Mexico DPS, the Central and Southwest Atlantic DPS continues to see heavy fishing pressure by commercial fisheries off the coast of Brazil and by artisanal fisheries in Central America, the Caribbean, and Brazil.

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 (PLL) 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 metric tons (mt) in 1999 to 508 mt by 2002, before decreasing to a low of 87 mt in 2009. Information from PLL 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 in some countries outside the United States 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).

3.2.5 Oceanic Whitetip Shark

On January 30, 2018, NMFS published a final rule to list the oceanic whitetip shark (*Carcharhinus longimanus*) as a threatened species under the ESA, effective March 1, 2018 (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
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. obscuras*), 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).

Distribution and Habitat Use

A geographical representation of the species range is provided by Last and Stevens (2009) (Figure 1). The oceanic whitetip shark is distributed worldwide in epipelagic tropical and subtropical waters between 30º North latitude and 35º South latitude (Baum et al. 2006). 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). 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 of shark that is usually found offshore in the open ocean, on the outer continental shelf, or around oceanic islands in deep water, occurring from the surface to at least 152 meters (m) depth. Essential fish habitat (EFH) for the oceanic whitetip shark includes localized areas in the central Gulf of Mexico and Florida Keys, and depths greater than 200 m in the Atlantic (from southern New England to Florida, Puerto Rico and the U.S. Virgin Islands. 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). The thermal preferences of oceanic whitetip sharks in conjunction with their reported range within 30° N and S suggest possible thermal barriers to inter-ocean basin movements around the southern tips of Africa and South America (Bonfil *et al.,* 2008; Musyl *et al.,* 2011; Howey-Jordan *et al.,* 2013; Gaither *et al.,* 2015).

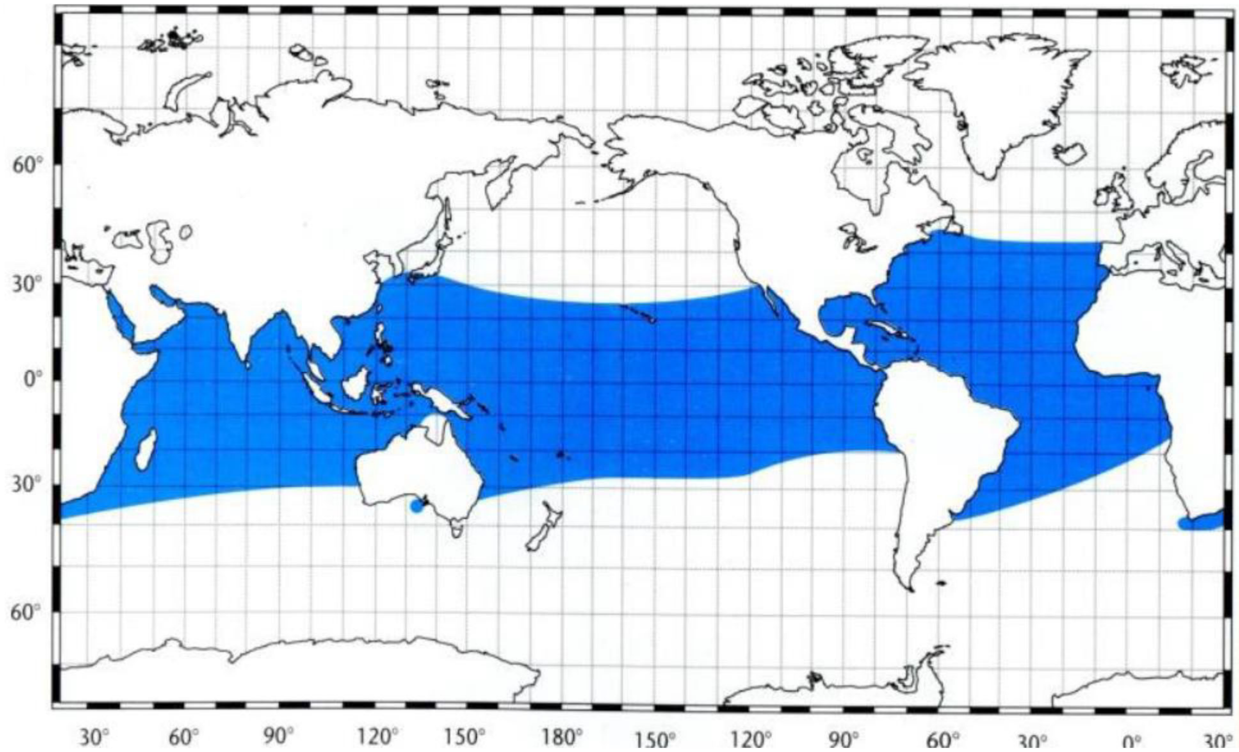

Figure 3.12 Geographic distribution of oceanic whitetip shark (Last and Stevens 2009)

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.

In the Atlantic Ocean, participants in the NMFS Cooperative Shark Tagging Program (CSTP) tagged 643 oceanic whitetips between 1962 and 2013, but only 8 were recaptured. Maximum time at liberty was 3.3 years, maximum distance traveled was 1,226 nmi (2,270 km), and maximum estimated speed was 17 nmi/day (32 km/day) (Kohler *et al.,* 1998; Kohler and Turner 2019). These data show movements by juveniles from a variety of locations, including from the northeastern Gulf of Mexico to the East Coast of Florida, from the Mid-Atlantic Bight to southern Cuba, from the Lesser Antilles west into the central Caribbean Sea, from east to west along the equatorial Atlantic, and from off southern Brazil in a northeasterly direction (Kohler *et al.,* 1998; Bonfil *et al.,* 2008); see Figure 3.13). An immature female was also tagged in the waters between Cuba and Haiti and was recaptured the next day within 6 nmi (11 km) of the tagging location (NMFS unpublished data; see Figure 3.13). Additionally, an adult of unknown sex was tagged and recaptured three years apart in the vicinity of Cat Island, Bahamas (NMFS unpublished data; see Figure 3.13 below).

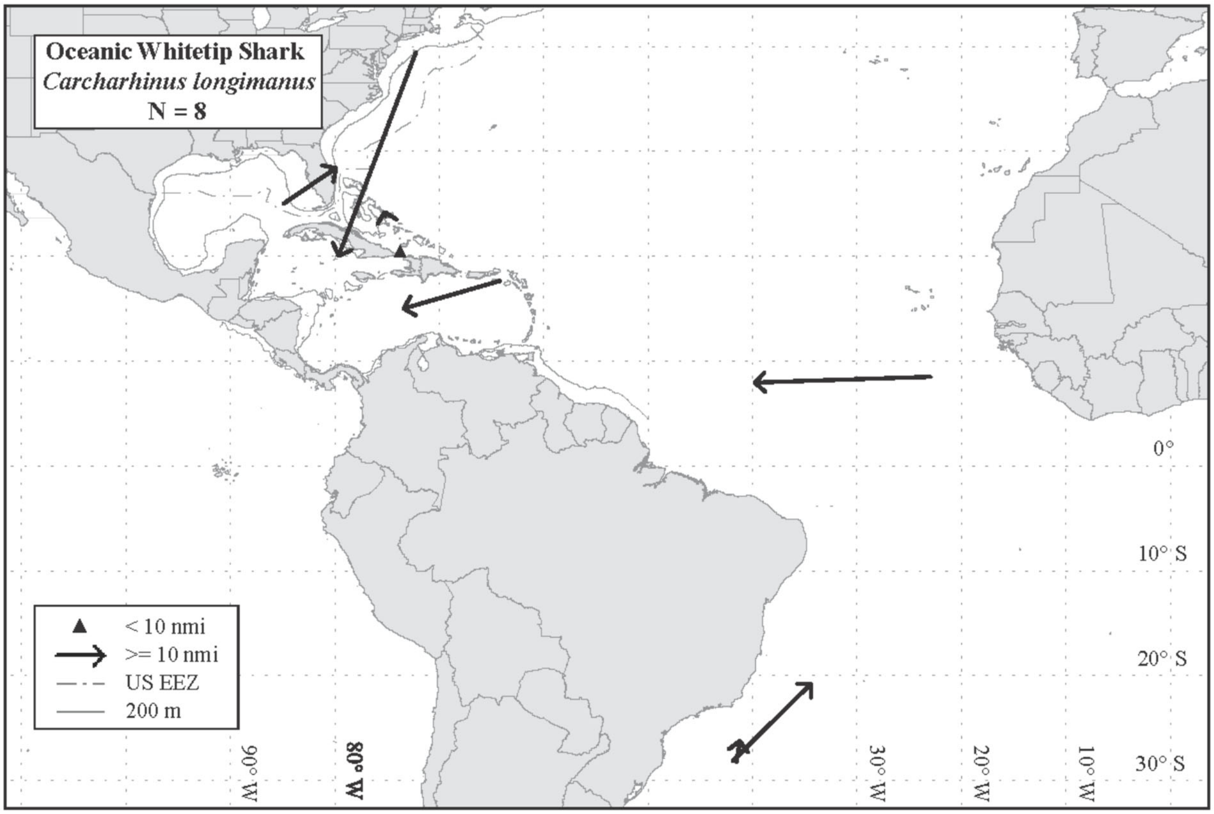

Figure 3.13 Recapture distribution for the oceanic whitetip shark from the NMFS Co-Operative Shark Tagging Program during 1962-1993 and NMFS Unpublished Data.

In the Gulf of Mexico, a satellite tagged oceanic whitetip shark moved a straight-line distance of 238 km from southeast Louisiana to the edge of the continental shelf about 300 km north of the Yucatan Peninsula. During the track, the shark rarely dove below 150 m staying above the thermocline, and only one dive to 256 m was recorded. The most frequently occupied depth during the entire track was 25.5-50 m (49.8% total time) and temperature was 24.05-26 °C (44.7% total time) (Carlson and Gulak 2012). More recently, a study from Cat Island, Bahamas tagged and tracked 11 mature oceanic whitetip sharks (10 females, 1 male). Individuals tagged at Cat Island stayed within 500 km of the tagging site for ~30 days before scattering across 16,422 km2 of the western North Atlantic (Howey-Jordan *et al*. 2013). Times at liberty ranged from 30-245 days, after which the largest movement by an individual from the tagging site ranged from 290–1,940 km. Individuals moved to several different destinations thereafter (e.g., the northern Lesser Antilles, the northern Bahamas, and north of the Windward Passage (the strait between Cuba and Haiti)), with many returning to the Bahamas after ~150 days. Howey-Jordan *et al.* (2013) found generally high residency times of oceanic whitetips in the Bahamas Exclusive Economic Zone (mean = 68.2% of time). Similar to the tagging study in the Pacific by Musyl *et al.,* (2011), oceanic whitetip sharks in the Bahamas spent 99.7% of their time in waters shallower than 200 m and did not show differences mean depths between day and night, with average day and night temperatures of 26.26±0.003oC and 26.23±0.003oC, respectively. According to Howey-Jordan et al. (2013):

"There was a positive correlation between daily sea surface temperature (SST) and mean depth occupied (i.e., as individuals experienced warmer SST, likely resulting from seasonal sea surface warming or migration to areas with warmer SST, mean daily depth increased, suggesting possible behavioral thermoregulation. All individuals made short duration (mean=13.06 minutes) dives into the mesopelagic zone (down to 1,082 m and 7.75◦C), which occurred significantly more often at night."

These tracking data also suggest that oceanic whitetip sharks exhibit site fidelity to Cat Island, Bahamas, although the reasons for this are still unclear. NMFS CSTP data (discussed earlier) from an adult oceanic whitetip, tagged and recaptured three years later in this area, provides supporting evidence of site fidelity to the waters around Cat Island. This information is important given the characterization of this species as highly migratory (Howey-Jordan *et al.,* 2013) (Figure 3.14).

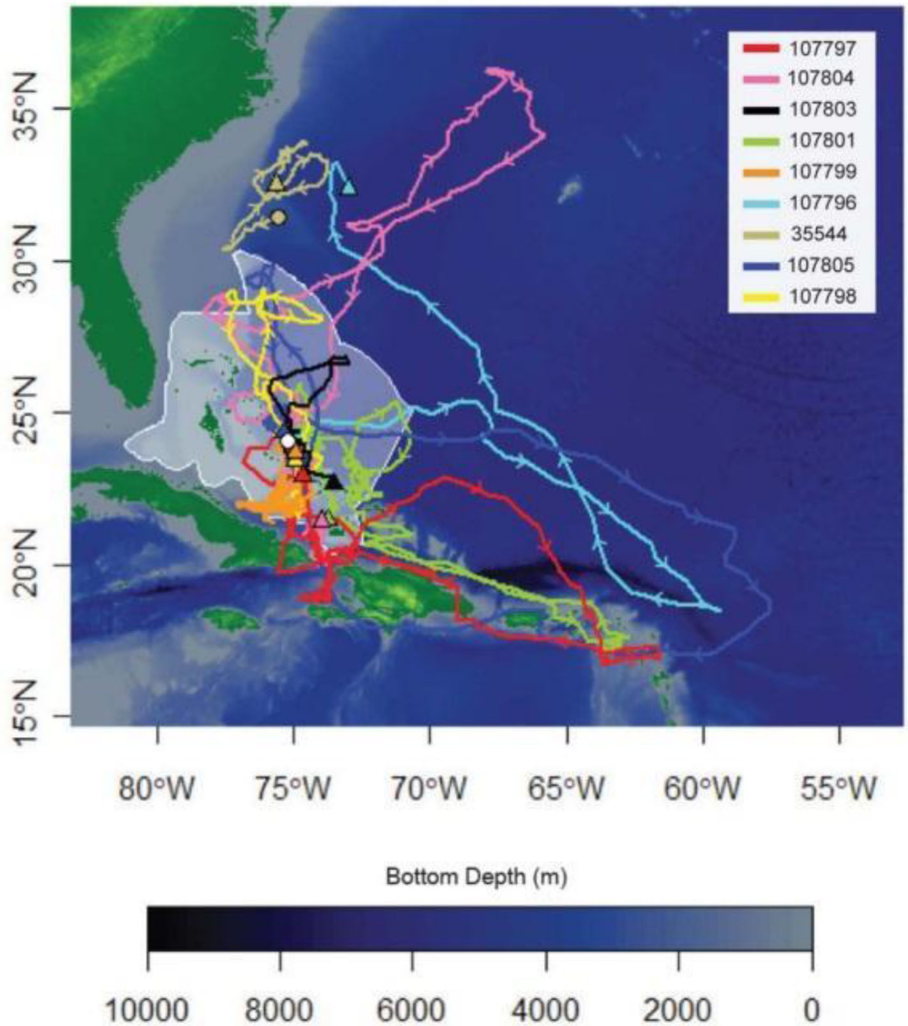

Figure 1.14 Map with bottom depth showing filtered tracks for nine oceanic whitetip sharks equipped with Standard Rate tags. Colored lines represent tracks from individuals (listed by tag ID) (Howey-Jorden et al. 2013)

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 long lived species (at least 20 years) and can be characterized as having relatively low productivity. Elasmobranchs evolved “slow” life history traits (e.g. late maturity, long gestation, slow growth, *K*-selected strategy) and behaviors (e.g. sex or age-specific migration and schooling, pronounced diel vertical diving patterns) that make populations vulnerable to exploitation and ultimately stock depletion, collapse and possible extinction (Baum, Myers, Kehler, Worm & Harley, 2003; Dulvy et al.,2008; Holden, 1973; Stevens, 2000).

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 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 and the ongoing threat of overutilization, 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; Dec. 29, 2016; see also 83 FR 4153 (Jan. 30, 2018)).

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., sperm whales, sea turtles, giant manta ray, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark), their habitats, 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's 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 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 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 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

Given the large size of the action area various species life stages and associated behaviors occur in the action area and are exposed the various threats each face. The status of the species (including DPSs) in the action area, as well as the threats to them, are best reflected in their range-wide status and supported by the species accounts in Section 3 (Status of Species).

4.2 Factors Affecting Sperm Whales in the Action Area

The following analysis examines actions that may affect sperm whales and their environments specifically within the action area. Sperm whales 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. A more generalized discussion of impacts outside of the action area is incorporated as part of the overall status of the species as detailed in Status of Species section, above. Sperm whales utilize deep, open ocean areas, and thus most coastal activities and fisheries are not expected to directly impact them.

4.2.1 Federal Fisheries

Commercial and recreational fisheries in the Gulf of Mexico have interacted with sperm whales in the past, but interactions are rare. For all fisheries for which there is a fishery management plan (FMP) or for which any federal action is taken to manage that fishery, the impacts have been evaluated via section 7 consultation. Sperm whales have the potential to become entangled in fishing gear such as longlines or gillnets. While this species is less susceptible to threats posed by fishing gear than other more coastal cetaceans, there are reports of a sperm whale entanglement in pelagic longline gear within the Gulf of Mexico. Further, Thode et al. (2015) and Straley et al. (2015) used passive acoustic monitoring and decoy sound production to demonstrate that sperm whales may be attracted to the acoustic cues of fishing vessels for catch depredation, which could lead to gear entanglement.

The U.S. Atlantic pelagic longline fishery is the subject of this consultation, and its operation has affected and is part of the environmental baseline for sperm whales in the action area for this consultation. Pelagic Observer Program (POP) reports show two reported sperm whale interactions in the Gulf of Mexico since 1992 (2008 and 2015). The 2008 entanglement was released alive and deemed not-seriously injured (Garrison et al. 2009). The 2015 entanglement, although the animal was released alive, it was expected that due to the nature of the entanglement that there was a 75% likelihood that it was seriously injured (Garrison et al. 2017). Sperm whales have also been observed during hauling operations for longline fisheries in the southern hemisphere but there were no confirmed entanglements (Ashford et al. 1996; Nolan et al. 2000).

4.2.2 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)). Research permits authorizing take of sperm whales cover activities such as photographing (photo-identification), tissue sampling, and tagging. All 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, NMFS consults with itself to ensure that issuance of such permits can be carried out in compliance with Section 7 of the ESA.

4.2.3 Federal Military Activities

Potential sources of adverse effects in the action area include operations of the U.S. Department of Defense. The U.S. Navy (USN) conducts military readiness activities, which can be categorized as either training or testing exercises, throughout the action area. USN activities are likely to produce noise and harass sea turtles throughout the action area. Formal ESA Section 7 consultations on these activities concluded that although there is a potential from some USN activities to affect sperm whales, 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 sperm whales.[10]

4.2.4 Private and Commercial Vessel Operations

Private and commercial vessels, including fishing vessels in recreational, state, or federal fisheries, 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. However, no Opinion has determined that vessel traffic operations would adversely affect sperm whales.

4.2.5 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. 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. 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 sperm whale recovery. No specific information on the impacts of climate change to sperm whales is currently available.

[10] 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)].

4.2.6 Oil and Gas Exploration and Collection Activities

Oil and gas operations on the outer continental shelf that have been ongoing for more than 50 years involve a variety of activities that may adversely affect ESA-listed species in the action area. These activities and resulting impacts include vessels making supply deliveries, drilling operations, seismic surveys, fluid spills, oil spills and response, and oil platform removals.

Natural seeps provide the largest petroleum input to the offshore Gulf of Mexico, about 95 percent of the total. (Mitchell et al. 1999) estimated a range of 280,000-700,000 bbl per year (40,000-100,000 tonnes per year), with an average of 490,000 bbl (70,000 tonnes) for the northern Gulf of Mexico, excluding the Bay of Campeche. Using this estimate and assuming seep scales are proportional to surface area, the (NRC 2003) estimated annual seepage for the entire Gulf of Mexico at about 980,000 bbl (140,000 tonnes) per year, or about three times the estimated amount of oil spilled by the 1989 Exxon Valdez event (about 270,000 bbl) (SteynSteyn 2010) or a quarter of the amount released by the DWH event (4.9 million bbl of oil) (Lubchenco and Sutley 2010). As seepage is a natural occurrence, the rate of approximately 980,000 bbl (140,000 tonnes) per year is expected to remain unchanged into the foreseeable future.

Seismic exploration
Seismic exploration is an integral part of oil and gas discovery, development, and production in the Gulf of Mexico. Seismic surveys are routinely conducted in virtually all water depths, including the deep habitat of the sperm whales. NMFS considered the effects of seismic operations in a biological opinion issued to BOEM on its 2007–2012 OCS Gulf of Mexico program. That opinion concluded that seismic surveys, with BOEM-required mitigation, were likely to adversely affect sperm whales by harassment. Required protective measures can be found in the BOEM NTL 2016-G02 "Implementation of Seismic Survey Mitigation Measures and Protected Species Observer Program." Oil and gas activities are not permitted in the Flower Garden Banks National Marine Sanctuary, except for occasional G&G surveys that require approval to occur.

Lease Sales and Drilling
The sale of OCS leases in the Gulf of Mexico and the resulting exploration and development of these leases for oil and natural gas resources has affected the status of ESA-listed species in the action area. BOEM administers the OCSLA and authorizes the exploration and development of wells in Gulf leases. As technology has advanced over the past several decades, oil exploration and development has moved further offshore into deeper waters of the Gulf. The development of wells often involves additional activities such as the installation of platforms, pipelines, and other infrastructure. Once operational, a platform will generate a variety of wastes including a variety of effluents and emissions. Each of these wastes can contribute to the environmental baseline.

Additionally, although the release of oil is prohibited, accidental oil spills can occur from loss of well control and thus adversely affect sperm whales in the Gulf of Mexico. Previous biological opinions considered the effects resulting from the variety of actions associated with lease sales and development. For example, a 2007 opinion on the effects of the Five-Year Outer

Continental Shelf Oil and Gas Leasing Program (2007-2012) in the Central and Western Planning Areas of the Gulf of Mexico determined that oil and gas leasing may adversely affect sperm whales but was found not likely to jeopardize their continued existences. However, that opinion did not contemplate the effects of a disastrous blowout and resulting extremely large oil spill event. Consultation has been reinitiated in light of the Deepwater Horizon oil spill. The effects of the spill are detailed in the Status of the Species section for sperm whales above.

Oil Rig Removals
Both the USACE and BSEE permit the removal of oil rigs in the Gulf of Mexico. These removals often use explosives to sever associated pile structures that can impact a variety of species, including any ESA-listed species, in the action areas. The USACE oversees rig removals in state waters while BSEE permits these activities in federal waters of the OCS. The USACE consults with NMFS on a project-by-project basis for decommissioning activities that use explosives.

In regard to rig removals in federal waters, a formal ESA section 7 consultation was completed with BSEE in 2006 and in 2008 the ITS was amended following completion of an MMPA rule. That opinion found that the permitting of structure removals in the Gulf of Mexico was not likely to result in jeopardy for sperm whales. In addition to the Reasonable and Prudent Measures within the ITS, BOEM has also issued "Decommissioning Guidance for Wells and Platforms" (NTL 2010-G05) to inform lessees about mitigation and reporting requirements.

4.2.7 Marine Pollution

Marine Debris
The discharge of debris into the marine environment is a continuing threat to the status of species in the action area, regardless of whether the debris is discharged intentionally or accidentally. Marine debris may originate from a variety of sources, though specific origins of debris are difficult to identify. A 1991 report (GESAMP 1990) indicates that up to 80 percent of marine debris is considered land-based and a worldwide review of marine debris identifies plastic as the primary form (Derraik 2002). Debris can originate from a variety of marine industries including fishing, oil and gas, and shipping. Many of the plastics discharged to the sea can withstand years of saltwater exposure without disintegrating or dissolving.

Marine debris has the potential to impact protected species through ingestion or entanglement (Gregory 2009). Both of these effects could result in reduced feeding, reduced reproductive success, and potential injury, infection, or death. Sperm whale ingestion of marine debris is a concern, particularly because their suspected feeding behavior includes cruising along the bottom with their mouths open (Walker and Coe 1990). Ingested debris may block the digestive tract or remain in the stomach for extended periods, thereby reducing the feeding drive, causing ulcerations and injury to the stomach lining, or perhaps even providing a source of toxic chemicals (Laist 1987; Laist 1997). Weakened animals are then more susceptible to predators and disease and are also less fit to migrate or breed (Katsanevakis 2008; McCauley and Bjorndal 1999).

Chemical pollutants
Pollution from a variety of sources including atmospheric loading of pollutants such as PCBs, stormwater from coastal or river communities, and discharges from ships and industries may affect sperm whales in the action area. Sperm whales may be adversely affected by marine pollution originating from federal, state, or private activities, though little is known regarding the specific pollutants or the effects pollutants may have on individuals. Further, we are unaware of the possible long-term and trans-generational effects of exposure to pollutants. It is not known if high levels of heavy metals, PCBs, and organochlorines found in prey species accumulate with age and are transferred through nursing. Nevertheless, the accumulation of stable pollutants such as heavy metals, polycholorobiphenyls [PCBs], chlorinated pesticides [DDT, DDE, etc.], and polycyclic aromatic hydrocarbons [PAHs]) is of concern.

Oil spills
Oil spills are accidental and unpredictable events, but are a direct consequence of oil and gas development and production from oil and gas activities in the Gulf of Mexico. Oil releases can occur at any number of points during the exploration, development, production, and transport of oil. Any discharge of hydrocarbons into the environment is prohibited under U.S. law.

Deepwater Horizon
On April 20, 2010, while working on an exploratory well approximately 50 miles offshore Louisiana, the semi-submersible drilling rig Deepwater Horizon (DWH) experienced an explosion and fire. The rig subsequently sank and oil and natural gas began leaking into the Gulf of Mexico. Oil flowed for 86 days, until the well was capped on July 15, 2010. Millions of barrels of oil were released. Additionally, approximately 1.84 million gallons of chemical dispersant was applied both subsurface and on the surface to attempt to break down the oil. There is no question that the unprecedented DWH event and associated response activities (e.g., skimming, burning, and application of dispersants) have resulted in adverse effects on listed species and changed the baseline for the Gulf of Mexico ecosystem.

The investigation conducted under the National Resource Damage Assessment regulations under the Oil Pollution Act (33 USC §2701 et seq.) assessed natural resource damages stemming from the DWH oil spill. Specific impacts sperm whales was determined (Trustees 2016). The findings of this assessment provide details regarding impacts to the environmental baseline of listed species and critical habitats in the Gulf of Mexico and is summarized below and can be found at http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan. The unprecedented DWH spill and associated response activities (e.g., skimming, burning, and application of dispersants) resulted in adverse effects on sperm whales. Despite natural weathering processes over the years since the DWH, oil persists in some habitats where it continues to expose and impact resources in the northern Gulf of Mexico resulting in new baseline conditions (BOEM 2016; Trustees 2016).

Sperm whales could have been exposed to toxic oil components through inhalation, aspiration, ingestion, and dermal exposure. There were 19 observations of 33 sperm whales swimming in DWH surface oil or that had oil on their bodies (Diaz 2015 as cited in Trustees 2016). The effects of oil exposure include physical and toxicological damage to organ systems and tissues, reproductive failure, and death. Sperm whales suffered from multiple routes of exposure at the same time, over intermittent timeframes and at varying rates, doses, and chemical compositions of oil. The estimation of effects to sperm whales is largely based on observed impacts to bottlenose dolphins resulting from exposure to DWH oil. The DWH oil spill occurred in deep water sperm whale habitat. The same routes of internal oil exposure (ingestion, inhalation, and aspiration) would have occurred in sperm whales that have been shown to adversely affect coastal bottlenose dolphins. The surface oil and vapors at the surface were more concentrated offshore near the leaking well head that could have exposed sperm whales to high levels of contaminants between dives that were known to have occurred with dolphins.

Using the pre-spill abundance estimate of 1,635 sperm whales in the Gulf of Mexico (DWH Trustees 2015) and applying the expected effects from bottle dolphins to sperm whales, NOAA (2015) determined that 16 percent of the Gulf of Mexico population or about 262 whales were exposed to DWH oil. Thirty-five percent of those whales (or approximately 92 whales) were likely killed. In total, an estimated 6 percent of the Gulf of Mexico sperm whale population was killed. The initial exposure likely resulted in whale deaths later in time due to adrenal and lung disease as was observed in bottlenose dolphins. In addition to the sperm whale deaths, an estimated 46 percent of exposed females that survived suffered reproductive failure through aborted fetuses or early calf death. Thirty-seven percent of all exposed whales, including pregnant females, likely suffered adverse health consequences as a result of DWH oil exposure.

At the population level, the SWSS study (Jochens et al. 2008) reported the overall proportion of calves within the mixed groups of sperm whales prior to DWH to be 11 percent. The proportion of calves observed in the Gulf of Mexico was similar to those reported for other stable populations of sperm whales reported off the Seychelles Islands and Sri Lanka in which calves make up 9.8 percent and 12.6 percent of those populations, respectively (Whitehead et al. 1997). Chiquet et al. (2013) conducted a sensitivity analysis for sperm whales and concluded that even under the best case parameters for vital rates for the stable population of sperm whales in the Gulf of Mexico, the growth rate of the population is extremely slow (about 0.96 percent per year) as has been reported for other sperm whale populations with a stable age distribution (Whitehead and Mesnick 2003).

In an assessment of the long-term reproductive effects that DWH is having on the Gulf of Mexico sperm whale population, Trustees (2016) completed population modeling based on the mortalities associated with adverse health consequences of oil exposure and the reduced reproductive success in pregnant females. It is likely the number of females and calves in the population has been reduced. Sixteen percent of the sperm whale population was exposed to oil. Considering these effects at the population level in the Gulf of Mexico, DWH oil exposure resulted in a maximum population reduction of seven percent requiring 21 years to recover to the pre-spill population size. The effects of the 21-year recovery period are slowing the recovery of the sperm whale population in the Gulf of Mexico. At a more subtle, but still crucial, level, the summed negative effects of the DWH oil spill on the Gulf of Mexico ecosystem across resources, up and down the food web, and among habitats, will continue to impact sperm whales due to the long life of marine mammals and their strong dependence on a healthy ecosystem (Bossart 2011; Moore 2008; Reddy et al. 2001; Ross 2000; Wells et al. 2004).

4.3 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, hawksbill sea turtle, and olive ridley 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.

4.3.1 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.

4.3.1.1 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 date 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. Below we are providing information on the most recent consultation relevant to understanding the effect of the fishery on ESA-listed sea turtle species, as these are the consultations relevant to understanding the environmental baseline for the species. Where a formal consultation was reinitiated and revised in a manner that altered conclusions about effects to ESA-listed sea turtles, that reinitation is noted below. NMFS may have taken action to modify operation of the fisheries beyond those described below, but to the extent those actions did not alter expected effects to sea turtles from the fishery, those management actions are not described.

Southeastern Shrimp Trawl Fisheries
NMFS has prepared Opinions on shrimp trawling numerous times over the years (most recently 2012 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).[11] 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

[11] 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.

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. Subsequently, on December 20, 2019, NMFS published a final rule requiring all skimmer trawl vessels 40 ft and greater in length to use TEDs with 3-inch bar spacing or less, beginning on April 1, 2021 (84 FR 70048). A new consultation on the shrimp fishery including the new TED requirement is currently underway. The ITS in the April 2014 Opinion 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.

Atlantic HMS Pelagic Longline Fishery

The HMS pelagic longline fishery is the subject of this consultation, and its 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 (consultation history), past consultations on this fishery by NMFS have been conducted, with the impact to sea turtle species described in Sections 1 and 2 (Proposed Action). An ITS was provided authorizing takes. Appendix A reports the takes authorized for the fishery prior to completion of this consultation.

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

On January 10, 2020, NMFS issued an Opinion on the operation of Atlantic HMS fisheries (excluding the pelagic longline fishery)[12] as carried out under the 2006 Consolidated Atlantic HMS Fishery Management Plan (2006 Consolidated HMS FMP), as amended. The non-PLL HMS fisheries use a number of gear types that are known to interact with sea turtles, including gillnets, bottom longlines, and vertical lines. These fisheries have been in operation for an extended period of time, and have affected and are part of the environmental baseline for sea turtles in the action area for this consultation. Because of the varied nature of the non-PLL fisheries, impacts occur to a broader cross-section of sea turtle species and age classes than the PLL fishery. However, total estimated takes are lower than in the PLL fishery. The latest estimates for takes in the non-PLL HMS fisheries include 91 loggerheads (51 being lethal), 22 Kemp's ridleys (11 lethal), 46 N. Atlantic DPS green turtles (25 lethal), 3 S. Atlantic DPS green turtles (2 lethal), 7 leatherback turtles (4 lethal), and 2 hawksbill turtles (1 lethal). These takes are expected to occur over 3-year time periods, and were found to not jeopardize the continued existence of any of those species.

[12] 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.

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 with HMS permits 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 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 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 on of non-jeopardy conclusions and have low anticipated incidental take levels, which are reported in Appendix A.

4.3.1.2 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.

4.3.1.3 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 issued and carried out 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. At the time of the drafting of this Opinion there are 31 active scientific research permits directed toward sea turtles that are applicable to the action area. 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 action, 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.

4.3.1.4 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. The new South Atlantic Regional Biological Opinion (SARBO) (March 27, 2020) concluded that the proposed action would adversely affect, but not jeopardize the continued existence of 5 sea turtle species (NA DPS green, SA DPS green, Kemp's ridley, leatherback, and NWA DPS loggerhead sea turtles), 6 sturgeon species (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and SA DPSs of Atlantic sturgeon, and shortnose sturgeon), giant manta ray, smalltooth sawfish, Johnson's seagrass, and 5 coral species (elkhorn, staghorn, lobed star, mountainous star, and boulder star coral). Anticipated takes for SARBO are provided in Appendix B.

4.3.1.5 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.

4.3.2 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.

4.3.3 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 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.

4.3.4 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.

4.3.5 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. Skimmer trawls 40 feet long and greater also are required to utilize TEDs with a maximum 3-inch bar spacing beginning on April 1, 2021 (84 FR 70048; Dec. 20, 2019).

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)].

4.4 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 federal 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, Jan. 12, 2017), and the Final Listing Rule (83 FR 2916, Jan. 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. In areas where the species is not subject to fishing, population abundance may be stable (Miller and Klimovich 2017).

4.4.1 Federal Fisheries

Presently, there is only one Opinion evaluating the effects of federal actions on the giant manta ray in the Southeast Region, the 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). That Opinion concludes that a total of 9 giant manta rays will be captured by those HMS fisheries every 3 years, with no mortalities.

Insufficient data and information exist to specify how many animals are taken in various federal fisheries (beyond the HMS fisheries). At this time, the giant manta ray status report and the proposed and final listing rules 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, including the HMS fisheries subject to the opinion above and evaluated in this opinion, and the Gulf of Mexico reef fish bottom longline fishery. However, given the low estimates of bycatch in U.S. fisheries, impacts from this mortality on the species are likely to be minimal. Giant manta rays are not a federally managed species under any FMP.[13]

4.4.2 Marine Pollution

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.

4.4.3 Non-Federal 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.

4.4.4 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 Start 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.

[13] 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.

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 relatively 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), and the proposed (78 FR 20717, Apr. 5, 2013) and final (79 FR 38213, July 3, 2014) listing rules, serve as the best source of information for threats to the species associated with federal fisheries.

4.5.1 Federal Fisheries

Atlantic HMS- Pelagic Longline Fisheries for Swordfish and Tuna
Atlantic pelagic longline fisheries for swordfish and tuna (the subject of this consultation) are known to incidentally capture the Central and Southwest Atlantic DPS of scalloped hammerhead sharks (Miller et al. 2014). An analysis of observer data 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). The future effect of the HMS pelagic longline fishery on listed species, including the Central and Southwest Atlantic DPS of Atlantic sturgeon, is the subject of this Opinion. Information on estimated impacts in the future are presented later in this document.

HMS-Atlantic Commercial and Recreational Fisheries for Shark, Swordfish, Tuna, and Billfish
In addition to the HMS PLL fishery, the other HMS Atlantic commercial and recreational fisheries for shark, swordfish, tuna, and billfish are also known to interact with scalloped hammerhead sharks. Some of the federally-managed fisheries for Atlantic HMS occur in the Caribbean. A January 10, 2020, Opinion on the non-PLL Atlantic HMS fisheries determined that 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. The Opinion estimated the take in those HMS fisheries would be no more than 7 scalloped hammerhead sharks from the Central and Southwest Atlantic DPS, with 4 mortalities, every 3 years.

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 scalloped hammerhead, is currently underway.

4.5.2 Fisheries Independent Monitoring

NMFS Integrated Fisheries Independent Monitoring (FIM) 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. No more than 1 lethal take is expected over continuing 5 year periods.

4.5.3 State or Private Actions

While the Final Listing Rule for 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.

4.5.4 Conservation and Recovery Actions Shaping the Environmental Baseline

Nationally, the U.S. has implemented two significant laws that specifically address 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, amending the High Seas Driftnet Fishing Moratorium Protection Act and the 2000 Shark Finning Prohibition Act provisions of the MSA to further improve domestic and international shark conservation measures, including additional measures against shark finning. Implemented by final rule on June 29, 2016 (81 FR 42285), the Act-- 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. U.S. exports of dried shark fins dropped significantly after the passage of the Shark Finning Prohibition Act in 2000. 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 United States' participation is small. Overall, the United States exports approximately 1 percent of all globally traded shark fins, and imports an even smaller percentage.. 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 and territories that have enacted laws addressing the possession, sale, trade, or distribution of shark fins, 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 96304, Dec. 29, 2016) and the final listing rule (83 FR 4153, Jan. 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 Fisheries

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. The effects of the HMS pelagic longline fishery in the U.S. Atlantic are assessed in this Opinion.

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 488 mt dressed weight (50 CFR 635.27(b)(1)(iii)(D)). 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.

Presently, there is one other Opinion evaluating the effects of federal actions on the oceanic whitetip shark in the action area, the January 10, 2020, Section 7 Consultation on the Operation of the HMS Fisheries (Excluding Pelagic Longline) under the Consolidated Atlantic HMS Fishery Management Plan. That Opinion analyzed the impacts of all U.S. HMS fisheries other than the pelagic longline fishery. That Opinion estimated the total 3-year take in the HMS non-PLL fisheries would be no more than 6 oceanic whitetip sharks that would result in 3 mortalities.

4.6.2 State and Private Actions

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). It should be noted that oceanic whitetip sharks are still occasionally caught as bycatch and landed in the action area despite its prohibited status when caught in association with ICCAT fisheries (NMFS 2012; 2014), as retention is permitted in other HMS 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 caught in association with ICCAT fisheries, 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, to no landings in 2015-2017 (NMFS 2012a; NMFS 2014a, NMFS 2018). 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, while in 2017 the numbers were very similar, with a total of 68 interactions, 81% (55 individuals) released alive, 16% (11 individuals) discarded dead, and 1.5% each (1 individual of each) reported unknown and lost at surface (NMFS 2014; 2015; 2018).

While the prohibition 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). In addition to general commercial and recreational fishing regulations for management of HMS, the United States has implemented two significant laws that specifically address the conservation and management of sharks: the Shark Finning Prohibition Act and the Shark Conservation Act, described above. States have also enacted shark finning bans. The 2017 Shark Finning Report to Congress lists states and territories that have enacted laws addressing the possession, sale, trade, or distribution of shark fins, 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). 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 United States' participation is small. Overall, the United States exports approximately 1 percent of all globally traded shark fins, and imports an even smaller percentage..

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 for pelagic longline gear, and finning regulations, contribute to the conservation 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, the U.S. catch constitutes only a small portion of global catch. Thus, regulatory mechanisms on the global stage (i.e., in regional fisheries management organizations), such as the previously discussed ICCAT oceanic whitetip measure (Rec. 10-07), are critical to effectively conserve the species.

5.0 Effects of the Proposed Action

In this section of our Opinion, we assess the effects of proposed operation of the HMS PLL fishery on listed species that are likely to be adversely affected by this proposed action. 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 PLL fishery 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 sperm whales, sea turtles, or giant manta rays. The prey/foraging base for oceanic whitetip and scalloped hammerhead sharks includes other pelagic predators that are caught with PLL including tunas, dolphin, and wahoo. As apex predators, the biomass of oceanic whitetip shark and scalloped hammerhead populations is normally much smaller than the available forage biomass. Atlantic tunas are managed internationally by ICCAT to achieve maximum sustainable yield. Under that goal, if stocks targeted by the fishery are not overfished, then the impacts from PLL fishing would not impact the availability of forage for oceanic whitetip or scalloped hammerhead sharks. If a forage species stock targeted by the PLL fishery is overfished, then availability of that forage species as prey for oceanic whitetip or scalloped hammerhead shares could be diminished. However, oceanic whitetip and scalloped hammerhead sharks prey on multiple species and it is likely that the stocks of another prey species may be healthy and provide forage while an overfished prey species may be rebuilding from being overfished. It is also important to keep in mind that the U.S. PLL fishery represents a small portion of the overall PLL fishing effort in the Atlantic Ocean and any impacts of the U.S. fishery on the forage base of oceanic whitetip or scalloped hammerhead sharks can be expected to be similarly small in proportion and possibly negligible. 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 associated with the proposed action are likely to adversely affect sperm whales, sea turtles, giant manta rays, scalloped hammerhead sharks, or oceanic whitetip sharks 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 PLL fishery 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.

In conducting this consultation, we searched all available databases for all listed species interactions in HMS PLL gear. This section details the information on interactions that have been documented for sperm whales, sea turtles, giant manta rays, scalloped hammerhead sharks, or oceanic whitetip sharks. Because the HMS PLL fishery underwent changes in gear type and fishery practices following the 2004 Opinion (see Sections 1 and 2) when possible, we used data since 2005 to evaluate the likelihood of listed species interactions in the fishery as it best represents the fishery as it is prosecuted today and is expected to be prosecuted in future years. We have no reason to believe the HMS PLL fishery take rates and effort levels analyzed in this Opinion will substantially change 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 provides more detailed information on fishing effort. We conservatively rounded any fractional numbers up to the next whole number to represent the next whole individual.

The other potential route of 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, and thus have more potential to strike a vulnerable species than during active fishing, though such interactions are more likely with smaller, recreational vessels not in use in the PLL fishery. We do not believe sperm whales, sea turtles, giant manta rays, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, or oceanic whitetip sharks are likely to be adversely affected by vessel interactions, as described below.

Regulations implementing section 7(a)(2) of the ESA require biological opinions to evaluate the effects of federal actions on ESA-listed species to determine if it would be reasonable to expect them to appreciably reduce listed species' likelihood of surviving and recovering in the wild by reducing their reproduction, numbers, or distribution (16 U.S.C. 1536; 50 CFR 402.02). The term “species” includes any subspecies of fish or wildlife or plants, and any distinct population segment (DPS) of any species of vertebrate fish or wildlife which interbreeds when mature.

5.1 Effects on Sperm Whales

Effects of the operation of the HMS PLL fishery analyzed in this Opinion on sperm whales stem from interactions with fishing gear. As discussed in Section 3.0, although sperm whales could interact with vessels in the HMS PLL fishery, those interactions are extremely unlikely to occur and this potential route of effect is discountable.

5.1.1 Types of Interactions and General Effects to Sperm Whales from HMS PLL Gear

Based on the configuration of pelagic longline fishing gear, the overlap of the fishery with sperm whale habitat, and past reports of sperm whale interactions with pelagic longline gear, we believe that this fishery is likely to adversely affect sperm whales. Fishing gear with line in the water column has the potential to entangle a whale (Johnson et al. 2005), and the greater the amount of time the line remains vertically extended through the water column, the greater the probability of encountering a whale becomes. When a whale encounters a line/hook it may become caught in the whale’s mouth, a pectoral fin, tail fluke, or wrap around the body. When the animal feels the resistance of the gear, it may thrash, which may cause it to become further entangled in the gear.

If the gear attached to the line is too heavy for the whale, it may drown. But whales have been observed swimming with portions of longline, with or without additional fishing gear, wrapped around a pectoral fin, the tail stock, or the mouth. Entangled animals may travel for extended periods of time and over long distances before freeing themselves, being disentangled by humans, or dying as a result of the entanglement (Angliss and Demaster 1998; Waring et al. 2013).

Entanglement may lead to exhaustion and starvation due to increased drag (Wallace 1985). Entanglements may also result in systemic infection or debilitation from tissue damage. Additionally, any injury or entanglement that restricts sperm whale from proper jaw movement while feeding, or prevents it from swimming at speeds necessary to capture prey will reduce its foraging capabilities and may lead to starvation (Cassoff et al. 2011; van der Hoop et al. 2012). A sustained stress response, such as repeated or prolonged entanglement in gear, makes marine mammals less able to fight infection or disease, and may make them more prone to ship strikes.

5.1.2 Factors Affecting the Likelihood of Sperm Whale Exposure to HMS PLL Gear

A variety of factors may affect the likelihood and frequency of sperm whales interacting with pelagic longline gear. The spatial and temporal overlap between fishing effort and sperm whale abundance as well as sperm whale behavior may be the most evident variables 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 and fishing techniques employed. However, there is little specific information available given the low frequency of observed sperm whale hooking or entanglement on pelagic longlines. These factors and their potential influence is discussed briefly below.

Spatial/Temporal Overlap of Fishing Effort and Sperm Whales
The likelihood and rate of sperm whale hookings and/or entanglements in HMS PLL gear is likely at least in part a function of the spatial and temporal overlap of sperm whales and fishing effort. The more abundant the whales are in a given area where and when fishing occurs, and the more fishing effort in that given area, the greater the probability a sperm whale will interact with gear. Environmental conditions, especially oceanographic features and fronts, may play a large part in where sperm whales are located in the action area, where the fishery's target species occur, and whether a whale interacts with the gear.

Hook Type
While entanglement appears to be the primary means of interaction between sperm whales and HMS PLL gear, the hook type may play a role. This fishery no longer uses J-hooks, so this may help reduce the likelihood of foul hooking a sperm whale if the gangion runs along the whale as it passes by a HMS PLL set. 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 sperm whale's skin.

Soak Time/Number of Hooks
Pelagic longline gear interactions with sperm whales 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 sperm whale may encounter the gear, either in attempts at depredation of fish caught on the gear or in simply passing across the gear during normal movements. Gear left in the water longer likely also has a greater chance of having more fish captured on the lines, and therefore a greater source of attraction for sperm whales attempting to depredate captured fish.

5.1.3 Estimating Interactions and Mortality of Sperm Whales in HMS PLL Gear

Minimal information is available regarding sperm whale takes from which to calculate an estimate of future interactions. Pelagic Observer Program records show only two recorded, observed interactions from 1992 until 2018 in the HMS PLL fishery. Both of those observations came in relatively recent years and both were in the Gulf of Mexico, which may be an indicator of the possibly increasing sperm whale population. In 2008, one individual, with a calf, was reported to have been entangled but was released alive and deemed not-seriously injured (Garrison et al. 2009). In 2015, a second individual was reported entangled around the tail/flukes in mainline and gangions, and though gear was partially removed, the animal was released with some gear still entangling it (Garrison et al. 2017). Therefore, due to the nature of the entanglement it was deemed that there was a 75% likelihood that it was seriously injured (Garrison et al 2017). Based upon observer coverage and fishery effort, the NMFS SEFSC calculated that the sperm whale take in 2008 resulted in an estimated 1.6 interactions with sperm whales that year, and the 2015 take calculation came out to 1.26 interactions with sperm whales that year. However, given the small sample size, the confidence intervals were very wide (2008: 0.30-8; 2015: 0.25-6.44), and the limited available data does not allow for a robust estimation of take. All of the information on sperm whale, and other sea turtle and marine mammal, interactions with the HMS PLL fishery can be found in the annual NOAA Technical Memoranda, prepared by the SEFSC, titled "Estimated Bycatch of Marine Mammals and Sea Turtles in the U.S. Atlantic Pelagic Longline Fleet during [Year]." Those memoranda describe the methods for extrapolating take to the fishery. In brief, the official annual estimates of take (total interactions and dead-on-retrieval) are based upon a SEFSC review of data from the pelagic observer program on observed interactions and observer coverage as well as data from the mandatory fishery logbook reporting program on fishing effort. They extrapolate both region-specific and total annual interactions with protected species in the fishery. Technical memoranda for the years 1992-2014, can be found in the searchable NOAA Institutional Repository: https://repository.library.noaa.gov/welcome. The Technical Memorandum for the year 2015 is on file with SERO. For 2016-2018, no technical memoranda were published, but the SEFSC conducted the same analyses using the same methods [SEFSC unpublished data]. There were no recorded, observed interactions in those years.

Given the limited available data and our desire to be consistent with the methodologies used to develop MMPA Stock Assessment Reports (SARS) and to conduct negligible impact determinations we looked at the most recent 5 years of available data (2014-2018) for take in the HMS PLL fishery. This time period provides a reasonable estimate of potential interactions and is more conservative than using a longer time series given the infrequency of observed interactions. As discussed above, in 2015 there was one observed interaction, which calculated out to an estimated 1.26 individuals interacting with the fishery that year (0.25/year). The past observed interactions have occurred in the GOM, where there is a greater degree of overlap between sperm whale concentrations and the fishery compared to the rest of the action area, and therefore we expect the estimated takes to occur in the GOM portion of the HMS PLL fishery, impacting the northern GOM stock of sperm whales. Because of inherent variability in interaction frequency, with many years having no takes, the overall rarity of an observed sperm whale take in the HMS PLL fishery, and the possibility of more than one take in a year, we believe it is appropriate to consider take levels over a 10-year rolling period. Estimating over a period of that length will help account for the variability, better reflects the extended time frames expected between interactions, and prevents the need to reinitiate consultation which would likely occur from just one observed take analyzed over a shorter time frame. We recognize that not all interactions with the HMS PLL fishery would be lethal. However, given the rarity of interactions and the potential for lethal interactions, we take a conservative approach and analyze the impacts as if all will be lethal for our jeopardy analysis. Therefore, we expect a total of up to 3 takes over any 10-year period (1.26 takes over 5 years = 2.52 takes over 10 years, rounded up to the next whole individual), in any combination of lethal or non-lethal.

5.2 Effects on Sea Turtles

Based on gear characteristics, fishing practices, sea turtle biology, and information on past interactions, we believe that the HMS PLL fishery may affect, and is likely to adversely affect, sea turtles. This section focuses on evaluating the effects of the HMS PLL fishery on sea turtles. The gear used by the fishery presents a significant threat to sea turtles.

Vessel traffic also has the potential to adversely affect listed species by direct strikes (i.e., impact with the vessel's hull or running gear), particularly those species that spend a significant amount of time near the surface like sea turtles. Although this threat has been analyzed in some coastal areas where there is significant overlap of sea turtles and vessel traffic (e.g., Florida), we do not have a general analytic tool to analyze the potential for vessel strikes in offshore, ocean environments given the general lack of stranding data and vessel traffic patterns, as well as the widespread nature of open pelagic waters compared to coastal areas. Lacking a more suitable analytical tool for vessel strikes by the HMS PLL fishery, we use Barnette (2018), which analyzed the probability of vessel strikes on sea turtles in coastal Florida waters, as a proxy in this Opinion. This represents a highly conservative (erring on the side of the species) analysis as sea turtles, as well as vessel activities, are much more concentrated in coastal areas compared to open waters where the HMS PLL fishery operates. While the vessels participating in the HMS PLL fishery must transit through coastal areas as they are leaving to fish or returning from fishing, the majority of their trip is spent in open ocean waters where sea turtles are much less concentrated.

Based on documented stranding data and vessel use patterns for various counties, Barnette (2018) estimated a vessel strike every 4,577-8,500 vessel trips under the most conservative of approaches (i.e., assuming low stranding returns and low number of annual vessel trips) and a vessel strike every 135,501 trips in areas with good stranding data and a high number of annual vessel trips. The HMS PLL fleet in the Atlantic and GOM currently has approximately 198 total vessels (30-145 ft in length, averaging roughly 55 ft) (HMS Management Division data). Between 2015-2018, the number of total HMS PLL trips in a year ranged from 921 to 1,185 (HMS Management Division data). Using even the most conservative estimate from Barnette (2018) (a vessel strike every 4,577-8,500 vessel trips), which is based on coastal areas and includes the impact of fast, planing recreational vessels in addition to larger commercial vessels, the number of trips by the HMS PLL fishery falls far below the threshold for one interaction in a year. Given that analysis, and the fact that the vast majority of each HMS PLL trip occurs on open waters with much lower sea turtle densities, we believe that the risk of sea turtles being struck by a vessel associated with the actions covered under this Opinion is extremely unlikely. Therefore, vessel traffic from the HMS PLL fishery may affect, but is not likely to adversely affect, sea turtles.

5.2.1 Types of Interactions and General Effects from HMS PLL Gear

Pelagic longline gear is 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 pelagic longline gear.

Entanglement
Sea turtles are particularly prone to entanglement as a result of their body configuration and behavior. Records of stranded or entangled sea turtles reveal that fishing gear can wrap around the neck, flipper, or body of a sea turtle and severely restrict swimming or feeding. If the sea turtle is entangled when young, the fishing line will become tighter and more constricting as the sea turtle grows, cutting off blood flow and causing deep gashes, some severe enough to remove an appendage.

Pelagic longline gear is fluid and drifts according to oceanographic conditions, including wind and waves, surface and subsurface currents, etc.; therefore, depending on sea turtle behavior, environmental conditions, and location of the set, turtles can become entangled in longline gear. Sea turtles have been found entangled in gangions, mainlines and floatlines. Sea turtles entangled in the longline fishery are most often entangled around the neck and foreflippers, and, in the case of leatherback turtles, are often found snarled in mainlines, floatlines, and gangions (e.g.*,* Hoey 2000). If sea turtles become entangled in monofilament line (mainline, gangion or float line), the gear can inflict serious wounds, including cuts, constriction, or bleeding anywhere on a turtle's body. In addition, entangling gear can interfere with a turtle's ability to swim or impair its feeding, breeding, or migration and can force the turtle to remain submerged, causing it to drown.

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 gastro-intestinal tract (Balazs et al. 1995). Observer data from the pelagic longline fishery 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. Following the 2004 Opinion and rules that required the fishery to use larger circle hooks instead of the previous standard J-hook, swallowing of hooks, and internal hooking, has been reduced.

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

Sea turtles can be forcibly submerged by longline gear. Forcible submergence may occur through a hooking or entanglement event, where the turtle is unable to reach the surface to breathe. This can occur at any time during the set, including the setting and hauling of the gear. Forced submergence can occur when the sea turtle encounters a line too deep below the surface, or because the line is too heavy to be brought up to the surface by the swimming sea turtle. The RPA in the June 14, 2001, opinion specified that gangion length be at least 110% of floatline length on shallow longline sets, which was adopted July 2, 2002 (67 FR 45393). This requirement was intended to reduce or eliminate the threat to turtles presented in that situation.

When interacting with longline gear, hooked sea turtles will sometimes drag the clip, attached to the gangion, along the mainline. If this happens, the potential exists for a turtle to become entangled in an adjacent gangion, which may have another species hooked such as a shark, swordfish, or tuna. If a turtle were to drag the gangion against another gangion with a live animal attached, the likelihood of the turtle becoming entangled in the second gangion is greater. If the turtle becomes entangled in the gear, then the turtle may be prevented from reaching the surface. The potential also exists, if a turtle drags the gangion next to a float line, the turtle may wrap itself around the float line and become entangled.

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).

Although a low percentage of turtles that are captured by longline fishermen actually are reported dead, sea turtles can drown from being forcibly submerged. Such drowning may be either "wet" or "dry." With wet drowning, water enters the lungs, causing damage to the organs and/or causing asphyxiation, leading to death. In the case of dry drowning, a reflex spasm seals the lungs from both air and water. Before death due to drowning occurs, sea turtles may become comatose or unconscious. Studies have shown that sea turtles that are allowed time to stabilize after being forcibly submerged have a higher survival rate. This depends on the physiological condition of the turtle (e.g. overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g. sea surface temperature, wave action, etc.), and the nature of any sustained injuries at the time of submergence (NRC 1990).

5.2.2 Factors Potentially Affecting the Likelihood of Exposure of Sea Turtles to HMS PLL Gear

A variety of factors may affect the likelihood and frequency of listed sea turtle species interacting with pelagic longline gear. The spatial and temporal overlap between fishing effort and sea turtle abundance as well as sea turtle behavior may be the most evident variables 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 and fishing techniques employed. These factors and their 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 PLL 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, especially oceanographic features and fronts, may play a large part in both where sea turtles are located in the action area, where the fishery's target species occur, and whether a sea turtle interacts with the gear.

Hook Type
The type of hook (size and shape) used may also impact the probability and severity of interactions with sea turtles. As described previously, this fishery no longer uses J-hooks, which helps to reduce the incidence, and severity, of hooking. 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 the HMS PLL fishery, the larger 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). The larger the hook (18/0 vs. the smaller 16/0) and whether the point of the hood is offset or not can play a role in likelihood of hooking a sea turtle. A more detailed explanation of the different impacts determined for the various hook sizes can be found in the 2004 Opinion for this fishery (NMFS 2004).

Soak Time/Number of Hooks
Pelagic longline gear interactions with sea turtles may be affected by both soak time and the number of hooks fished, independent of overall fishing effort. The greater the soak time, the greater the chances a foraging sea turtle may encounter the gear, and the longer a sea turtle may be exposed to the entanglement or hooking threat, increasing the likelihood of such an event's occurrence.

Floats
Sea turtles may be attracted to the floats used on longline gear. According to a study by Arenas and Hall (1992), turtles show a preference for nearly submerged objects floating horizontally and are strongly attracted to brightly colored objects. Lab experiments have shown sea turtles prefer bright colors (i.e., red and yellow) over dull or darker colors (i.e. black, green or blue) (e.g. Fontaine, et al. 1985). Controlled experiments and qualitative evaluations were conducted by the SEFSC using captive reared sea turtles to evaluate their responses to various components of pelagic longline gear and other stimuli. One experiment tested the attraction of sea turtles to orange and white colored longline floats in an 80' x 35' pen enclosure. Sea turtles were introduced into the pen with a single float treatment. Preliminary analysis of the results indicated that the test turtles may have been more attracted to orange-colored floats than to white-colored floats (J. Watson, SEFSC, personal communication, July 2001). Floats typically used during swordfish- style sets are bright orange, bullet-shaped, and slightly submerged. Deep sets generally use larger cylindrical inflatable or rigid spherical buoys and floats, and

these also are typically orange in color (L. Enriquez, NMFS, personal communication, January 2001; e.g. www.lindgren-pitman.com/floats.htm).

Mainline and hardware
The SEFSC also conducted evaluations at the Panama City Laboratory that involved placing longline gear in open water pens with captive reared loggerhead turtles to investigate turtle entanglement with various longline gear components. During these experiments, scientists observed turtles tracking along the mainline and biting at the hardware (snaps). Turtles placed in a pool without longline gear (i.e., control) tended to track along the outside edges of the pool. These observations support at-sea observations by divers and remotely operated vehicles, which indicate that turtles may be attracted to the highly visible mainline and hardware used by the fishing industry, and that the turtles may swim along the mainline (J. Watson, SEFSC, personal communication, August 2001).

Lightsticks
Sea turtles foraging at night may be attracted to the lightsticks, confusing them for prey or simply investigating novel items in their environment. Lightsticks are often used by longliners targeting swordfish in order to attract the swordfish to the bait. Whether lightsticks attract swordfish directly or whether they attract baitfish, which in turn attract the swordfish, is not entirely clear; however, fishermen report higher takes of swordfish when they use lightsticks. Lightsticks are generally attached to every gangion, approximately a meter above the hook. Leatherback, loggerhead, and olive ridley turtles are known to prey on pyrosomes, common, naturally occurring animals known as “fiery bodies,” which suggests they may be attracted to lightsticks; however, there is little information on the ingestion of lightsticks by sea turtles. In addition, statisticians have not been able to find any correlation between sea turtle take and the proximity of a lightstick to the hook or branchline the turtle was hooked on or entangled in. Experimental studies have, however, indicated that juvenile sea turtles orient towards green, blue, and yellow chemical lightsticks, and orange, green, and shaded green battery powered LEDs (Wang, et al. 2004).

Bait
Sea turtles may also be attracted to the bait used on longline gear. 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. In the pelagic longline fishery, 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).

5.2.3 Estimating Sea Turtle Incidental Catch in U.S. Atlantic HMS PLL Gear

The proposed action is to generally continue status quo operation and management of the HMS PLL fishery, and so is not expected to increase effort within the fishery.[14] In addition, all of the gear restrictions enacted following the 2004 Opinion remain in effect. The fishery from 2005 (the first full year operating under the 2004 requirements) to 2018 is similar in scope and impact to what is expected to occur in the fishery in future years. We will therefore base future expected takes on past takes from 2005-2018. Including all of those years also allows us to better capture the inherent annual variability that occurs in interactions with sea turtles across the fishery. Any substantial changes to the fishery effort or practices in the future from the level across 2005-2018 would require a reexamination of the analysis and conclusions of this Opinion.

As presented in Table 1.1, in accordance with the requirements in the 2004 Opinion, and to track compliance with the ITS, NMFS calculated the total estimated incidental take levels for leatherback and loggerhead sea turtles for every 3-year period starting in 2004 (e.g., 2004-2006, 2007-2009, 2010-2012, 2013-2015, 2016-2018). Estimates from 2004-2015 are based on a series of annual NOAA Technical Memoranda, prepared by the SEFSC, titled "Estimated Bycatch of Marine Mammals and Sea Turtles in the U.S. Atlantic Pelagic Longline Fleet During [Year]." Those memoranda calculate the take based upon observer reports, observer coverage, and effort levels in the fishery. For 2016-2018, no technical memoranda were published, but the NMFS SEFSC conducted the same analyses using the same methods [SEFSC unpublished data]. A detailed explanation of the methodology used for the estimations can be found in each of the technical memoranda.[15] However, as stated above, the fishery did not fully implement the requirements of the 2004 Opinion until the 2005 fishing season, so we separated out the 2005 and 2006 take estimates in our summary of post-2005 estimated take levels (Table 5.1). The take estimates in Table 5.1 are based on the estimated actual interactions, not the take levels authorized in the ITS in the 2004 Opinion. Based on the total incidental take (across all years from 2005-2018), we estimate 332 leatherbacks will be incidentally taken per year, and 360 loggerheads will be incidentally taken per year by the HMS PLL fishery. We round up all results to the nearest whole animal for all calculations. Because there is high variability in take year to year, we use a 3-year estimate to help smooth out some of that variability. Therefore, we expect a total of 996 leatherback and 1,080 loggerhead incidental takes every three years.

Table 5.1 Estimated leatherback and loggerhead incidental takes from 2005-2018.

Year	Species	Estimated Total Incidental Take*
2005	Leatherback	351
	Loggerhead	274
2006	Leatherback	415
	Loggerhead	561

[14] As discussed in the Proposed Action section, the GRA/Weak hook rule, which re-opened two areas previously closed to PLL fishing (85 FR 18812), is not expected to increase effort. While the rule might redistribute effort, overall effort is likely to remain unchanged due to other regulatory restrictions in place for the fishery.

[15] The availability of the Technical Memoranda is discussed in section on effects to sperm whales, above.

2007-2009	Leatherback	1,167
	Loggerhead	1,557
2010-2012	Leatherback	1,007
	Loggerhead	1,464
2013-2015	Leatherback	947
	Loggerhead	882
2016-2018	Leatherback	753
	Loggerhead	294
Annual Average	Leatherback	**332**
	Loggerhead	**360**

*Note: estimated takes for 2005 and 2006 are single year totals, the subsequent take estimates are three-year combined totals.

It is notable that these numbers of expected total 3-year interactions (996 leatherback and 1,080 loggerheads) are substantially lower than those in the 2004 Opinion, which estimated 1,764 leatherback and 1,905 loggerhead interactions every 3 years, after the initial 2004-2006 time period, when the circle hook requirement went into place. The data on estimated incidental takes indicates a general decrease in total interactions over time despite the fishery operating under the same sea turtle protection regulations anticipated in the 2004 Opinion, including the circle hook requirement. Effort has varied annually and is expected to continue varying within a similar range in the future. Because of the variability in effort, we averaged over a 14 year time period (2005-2018). The reason for the decline in total interactions is unclear and could be the result of any one or combination of factors, from the fishers becoming better at avoiding sea turtles to changes in the populations of sea turtles in the size and age class that interacts with the HMS PLL fishery. While effort levels were lower in the 2016-2018 period compared to previous years, the difference in effort does not fully account for the low total interaction levels, and the basis for the difference is not well understood. It is possible that the very low loggerhead interactions in the 2016-2018 time period (294 interactions estimated over the 3-year period) is a result of a smaller cohort of the pelagic juveniles that interact with the fishery. Given their size, loggerheads typically interacting with the fishery are expected to be in the 8-12-year-old range. During the 2004-2009 seasons, loggerhead nesting had dipped to the lowest level at any time between 1989 and 2018 on the Florida nesting beaches, and it is individuals from that cohort that would be expected to have interacted with the HMS PLL fishery in the 2013-2015 and 2016-2018 time periods. If that is in fact the case, we may see interaction levels increase again from that low, as individuals from later cohorts during times of higher nesting levels (post-2009; see nesting information in Status of the Species section above) begin to reach the size and age that interacts with the HMS PLL fishery.

For hardshell turtles other than loggerheads, it is difficult to make predictions about future levels of interaction. Reported interactions are very low, often zero, sometimes one or two per year, observed. Additionally, there are likely misidentifications, as well as "unknowns" recorded in the observer data. Using the same bycatch data and analyses as for leatherback and loggerhead sea turtles (i.e., the NOAA Technical Memoranda, prepared by the SEFSC, and SEFSC analyses for the years 2016-2018), it was estimated that (again rounding all decimals up to the nearest whole turtle), from 2005-2018, a total of 16 green turtles, 12 Kemp's ridleys, 12 olive ridleys, and 58 "unidentified" turtles (12 dead on retrieval) were taken in the HMS PLL fishery. We also include hawksbill turtles in the assessment of potential "other hardshell" species that could interact with the fishery, as there are historical interaction records prior to 2005. Taking a conservative approach and assuming the “unidentified” turtles are “other hardshell” species, and not loggerhead sea turtles, we estimate that 98 “other hardshells” were taken over the 14-year period; therefore, we average 7 total “other hardshells” per year. Consistent with other species, we utilize a three-year period to reduce the effect of inherent variability when assessing whether the fishery has exceeded the expected level of take. Therefore, we estimate that the fishery may take up to 21 of any combination of the “other hardshell” turtle species, namely green, Kemp’s ridley, olive ridley, and hawksbill over three years. Later in this Opinion, when evaluating whether take of these species is likely to jeopardize their continued existence, we will take a conservative approach and assess the impact to green, Kemp’s ridley, olive ridley, and hawksbills as if each of the total 21 individuals taken over three years were of that species. We take this conservative approach because take for any of these species is highly variable, and we have “unidentified” individuals.

5.2.3 Estimating Sea Turtle Mortality from HMS PLL Gear

Because of the gear configuration and fishery characteristics, relatively few sea turtles captured on pelagic longlines are dead as a result of injury or forcible submergence when boated or released. In the 2004 Opinion, using observer data from 1992 until 2002, we estimated that only 1.1% of the total number of sea turtles (all species) are dead when brought on board. The result did not vary much if the data are separated into leatherbacks (1.3% dead) and hardshell turtles (1.0% dead). The low percentage of dead-on-retrieval individuals is further illustrated when looking at the more recent observer data. The total estimate of dead-on-retrieval leatherback and loggerhead sea turtles combined accounted for only 0.92% of the combined catch of those two species (108 total dead-on-retrieval/11,765 total takes) (Table 5.2). Breaking it down by species, dead-on-retrieval individuals accounted for 1.3% of the leatherback takes (77 dead-on-retrieval/5,999 takes), and 0.54% of the loggerhead takes (31 dead-on-retrieval/5,766 takes). The low percentage of sea turtles suffering immediate mortality as a result of fishing gear interactions is likely due to the various gear modifications enacted prior to 2004, including longer gangion lengths (allowing the turtles to surface more easily to breath, enacted in 2001) in addition to the later hook type changes (reducing the severity of the physical injury from hooking, starting in the second half of 2004).

Table 5.2 Estimated leatherback and loggerhead sea turtle impacts since the 2004 Opinion.

		Estimated Total Incidental Takes (2004 Opinion ITS Level)	Post-Release Mortality Rate (2004 Opinion RPA Level)	Estimated Post-Release Mortalities	Estimated Dead on Retrieval	Total Mortality (2004 Opinion RPA Level)	Total Mortality Exceedance?
2004-2006	Leatherback	2,125* (1,981)	22.5% (26.2% by Q1 2005; 19.6% by Q1 2006)	471	34	505 (548)	No
	Loggerhead	1,569 (1,869)	28.8% (20.2% by Q1 2005; 18.6% by Q1 2006)	452	0	452 (438)	Yes
2007-2009	Leatherback	1,167 (1,764)	24.3% (13.1%)	277	29	306 (252)	Yes
	Loggerhead	1,557 (1,905)	23.3% (17%)	360	12	372 (339)	Yes
2010-2012	Leatherback	1,007 (1,764)	22.1% (13.1%)	222	4	226 (252)	No
	Loggerhead	1,464 (1,905)	25.7% (17%)	377	0	377 (339)	Yes
2013 - 2015	Leatherback	947 (1,764)	30.1% (13.1%)	285	3	288 (252)	Yes
	Loggerhead	882 (1,905)	27.3% (17%)	238	12	250 (339)	No
2016 - 2018	Leatherback	753 (1,764)	35.2 %** (13.1%)	263	7	270 (252)	Yes
	Loggerhead	294 (1,905)	27.0** (17%)	78	7	85 (339)	No

*Over 47% of the estimated takes during 2004-2006 occurred prior to implementation of circle hooks in quarter 3 of 2004.
** 2018 mortality rate estimated using mean of 2016 – 2017.

Most, if not all, sea turtles released alive from HMS PLL gear will have experienced a 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 2006, NMFS revised the criteria for estimating post release mortality of sea turtles (Ryder et al. 2006). Under the revised criteria, overall post-release 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 when applying them to the data associated with the observer reports (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). The number of post-release mortalities are calculated by applying the post-release mortality rate, calculated from the observer information as described, to the number of individuals released alive. For example, between 2007 and 2009, NMFS estimated 1,167 interactions with leatherback sea turtles, of which 29 were estimated to have been dead on retrieval and 1,138 were released alive. Based on the estimated post-release mortality rate of 24.3%, NMFS estimates that 277 leatherback sea turtles suffered post-release mortality (1,138 * 24.3% = 276.5, rounded to 277).

In the 2004 Opinion, NMFS included in the Reasonable and Prudent Alternative a target for post-release mortality, shown in Table 5.2, in order to avoid jeopardy for leatherback sea turtles at the time. The observers are required to provide information on hooking location and amount of trailing gear left on every observed turtle that is released, in order to allow for estimates of post-release mortality in the fishery. The target level of post-release mortality was based on the rates achieved in the experimental fishery in the NED. As discussed in Section 1, the NED experimental fishery was established following the RPA in the 2001 Opinion. It is evident that the fishery has not been able to meet the post-release mortality rates specified in the RPA in the 2004 Opinion for leatherbacks or loggerhead sea turtles, with greater increases in the post-release mortality rate for leatherbacks in more recent years. The cause is not clear. As discussed, the post-release mortality rates are derived by applying the observer data on hooking location and gear removed prior to release of each turtle observed, to the revised post-release mortality table in NMFS SEFSC (NMFS 2012d). It is possible that the data would show that gear removal levels are not sufficient to reach the lower mortality rates or that hooking locations have changed, resulting in greater injuries to the species. Because the fishery requirements and effort levels are expected to remain similar in the coming years, the estimates are also representative of the impacts we can expect in future years. We evaluate whether the proposed action is likely to jeopardize the continued existence of these species based on the total expected future interactions and total expected future mortalities discussed here. If the post-release mortality rates stay the same in future, but the number of interactions increases, then total mortality will increase, and we would need to revisit the conclusions in this Opinion.

Table 5.3 Post-Release Mortality Rate in the HMS PLL fishery (2004-2017) vs. 2004 Opinion RPA levels. (Note: the average post-release mortality rate is calculated with data starting in 2005, as the 2004 season did not have the gear requirements for the entire fishing season).

	Leatherback		Loggerhead	
	Post-Release Mortality Rate	2004 Opinion RPA Level	Post-Release Mortality Rate	2004 Opinion RPA Level
2004	26.0	32.8	34.8	21.8
2005	15.4	26.2	23.6	20.2
2006	21.9	19.6	24.5	18.6
2007	25.1	13.1	20.9	17.0
2008	23.7	13.1	25.6	17.0
2009	28.3	13.1	23.0	17.0
2010	21.8	13.1	23.2	17.0
2011	22.4	13.1	28.1	17.0
2012	24.2	13.1	25.4	17.0
2013	25.9	13.1	25.4	17.0
2014	34.4	13.1	34.0	17.0
2015	31.9	13.1	25.2	17.0
2016	34.2	13.1	29.2	17.0
2017	37.1	13.1	23.0	17.0
Average	26.6		25.5	

Using the estimates for total interactions, average percent of individuals dead on retrieval, and average percent post-release mortality, we can estimate total mortality of leatherback and loggerhead sea turtles from the fishery in future years. To do so we use the following formulas:

total interactions x average dead-on-retrieval percentage = number dead on retrieval
total interactions – number dead on retrieval = number released alive
Total mortality = number dead on retrieval + (number released alive x average percent post release mortality)

For leatherback sea turtles we estimated 996 total interactions over each three-year period. We also calculated an average dead-on-retrieval percentage of 1.3%, and an average post release mortality of 26.6%. Therefore, we estimate that 275 leatherbacks will be killed by the fishery every three years.

996 x 1.3% = 13 leatherbacks dead on retrieval (12.949 rounded up to nearest whole turtle)
996 – 13 = 983 leatherbacks released alive
Total 3-year leatherback mortality = 13 + (983 x 26.6%) = 275 (274.5 rounded up to nearest whole turtle)

For loggerhead sea turtles we estimated 1,080 total interactions over each three-year period. We also calculated an average dead-on-retrieval percentage of 0.54%, and an average post release mortality of 25.5%. Therefore, we estimate that 280 loggerheads will be killed by the fishery every three years.

1,080 x 0.54% = 6 loggerheads dead on retrieval (5.8 rounded up to nearest whole turtle)
1,080 – 6 = 1,074 loggerheads released alive
Total 3-year loggerhead mortality = 6 + (1,074 x 25.5%) = 280 (279.87 rounded up to nearest whole turtle)

For "other hardshell" sea turtles, we continue the approach of combining data. While the limited data we have did not include any dead-on-retrieval takes for the other hardshell turtles identified to species, the "unidentified" turtles had 12 dead-on-retrieval individuals. Taking the total estimate of 98 "other hardshell" interactions, 12 dead-on-retrieval equals 12.2%. This percentage is much higher than for loggerhead sea turtles, a hardshell sea turtle, but we do not currently have any insights as to why. Because we do not have specific post-release mortality (based on gear removal and hooking location) for the "other hardshell" turtles, we use the average loggerhead post-release mortality percentage as a proxy. Therefore, we estimate that up to 8 non-loggerhead hardshell sea turtles will be killed by the fishery every three years.

21 x 12.2% = 3 "other hardshell" dead on retrieval (2.6 rounded up to nearest whole turtle)
21 – 3 = 18 "other hardshell" released alive
Total 3-year "other hardshell" mortality = 3 + (18 x 25.5%) = 8 (7.59 rounded up to nearest whole turtle)

5.3 Effects on Giant Manta Ray

We believe that the gear used in the HMS PLL fishery by commercial fishermen may adversely affect giant manta rays. This section focuses on evaluating the potential effects of pelagic longlines on giant manta rays.

We also believe that giant manta ray could be struck by a vessel associated with the actions covered under this Opinion, however that potential route of effect is extremely unlikely and discountable. While giant manta rays can be frequently observed traveling just below the surface and will often approach or show little fear toward vessels, few instances of confirmed or suspected strandings of giant manta ray are attributed to vessel strike injury. In general, information about interactions between vessels and giant manta rays is limited, particularly within, or near, the offshore areas where the HMS PLL fishery operates. This lack of documented mortalities could also be the result of other factors that influence carcass detection (e.g., wind, currents, scavenging, decomposition, etc.); however, giant manta rays appear able to move fast enough to avoid most moving vessels, as is anecdotally evidenced by videos showing high speed vessels passing over giant manta rays and the ray being able to avoid the interaction. Available information indicates the threat of vessel strike on giant manta ray is predominantly an issue in shallow, coastal waters and in proximity to inlets where giant manta ray frequent, likely to facilitate feeding. Due to the expected low concentration of animals in the action area, very limited reports of vessel interactions, and ability to avoid moving vessel traffic outside of confined spaces, we think it is extremely unlikely that vessels associated with the proposed action will encounter giant manta rays.

5.3.1 Types of Interactions and General Effects from HMS PLL Gear

HMS PLL gear would affect 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 also suffer impaired swimming (which can also impact water flow over their gills, and thus, respiration) or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns, though we have no actual evidence of such effects from this fishery. There is little available information on confirmed giant manta ray interactions with the HMS PLL fishery, however, there are rare reports of manta rays incidentally caught by the fishery that were not identified to species. The types of interactions would be expected to be similar for the various manta species, including the giant manta ray.

5.3.2 Factors Affecting the Likelihood of Giant Manta Ray Exposure to HMS PLL Gear

A variety of factors may affect the likelihood and frequency of giant manta rays interacting with pelagic longline gear. The spatial and temporal overlap between fishing effort and giant manta ray abundance as well as giant manta ray behavior may be the most evident variables involved in anticipating interactions. Other fishing related factors that may influence the likelihood and frequency of hooking and entanglement include gear characteristics and fishing techniques employed. These factors and their potential influence is discussed briefly below.

Spatial/Temporal Overlap of Fishing Effort and Giant Manta Rays
The location of the fishery in relation to the species is a factor influencing the likelihood that the HMS PLL fishery 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 but the pelagic longline fishery only occurs offshore.

Gear Usage and Fishing Techniques (Soak Times and Number of Hooks)
The amount of fishing effort would likely affect giant manta rays that are incidentally caught by the HMS fishery. Number of fishers, number of trips, number of hooks, 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.

5.3.3 Estimating Interactions with Giant Manta Rays in HMS PLL Gear

There is little data on giant manta ray interactions with the HMS PLL fishery from which to estimate future interactions. A review of the HMS PLL logbook data indicates that no giant manta rays have been reported caught from 1995 to 2018. However, logbook data do contain an occasional report of what were identified as manta ray, but with no identification to species. Likewise, little observer program data is available for giant manta ray captures in the fishery. Prior to 2019, observer reports did not have species-specific codes for any of the rays, and differentiation between mantas, or even other rays, was not reported. Given the addition of species-specific codes, future observer reports should provide better, more accurate identification. Therefore, until better data is available, the SEFSC (C. Jones, NMFS SEFSC Mississippi Laboratory) utilized the logbook and observer data as follows:

- The SEFSC used data from 2005-2018, because, prior to 2005, all rays regardless of species were lumped together in the observer data, whereas after 2005, observers had the option to report interactions as generic skate/ray or to differentiate “mantas” and “pelagic stingrays.” From 2005-2018, the generic “manta” designation could be used to identify mobulids (manta and devil rays). Beginning in 2004, the pelagic stingray designation was added, but that term was not widely or reliably used until 2005. Additionally, as explained previously, the year 2005 represents the first full year of the fishery operating under the current gear requirements.
- The SEFSC assumed that all observed “mantas” under 150 cm were not giant manta rays because newborn giant mantas are larger than 150 cm.
- The SEFSC conservatively assumed that all observed “mantas” above 350 cm were giant manta rays as mobulids in the action area above 350 cm are typically giant manta rays.
- The SEFSC assumed that anything in between 150-350 cm and designated as a generic skate/ray (i.e., not designated as a “manta”) was not a giant manta ray (which accounted for very few examples); this assumes that the observers would have used the “manta” designation if the interaction was with a mobulid (manta or devil ray) given that mobulids look notably different from other rays.
- After analyzing photos of “mantas” from the observer program, the SEFSC conservatively estimated that 13% of all “mantas” between 150-350 cm were giant manta rays. If a photo identification was uncertain but possibly giant manta ray, it was counted as a giant manta ray.

Based on that analysis, the SEFSC calculated 214.19 giant manta rays taken during observed sets in the fishery from 2005-2018 [SEFSC unpublished data]. This number is likely higher than expected in future years (and thus errs on the side of the species), as CPUEs have been generally declining over the past 6 years, indicating that average expected take per year would be lower for more recent years than for the longer time frame (Figure 5.1).

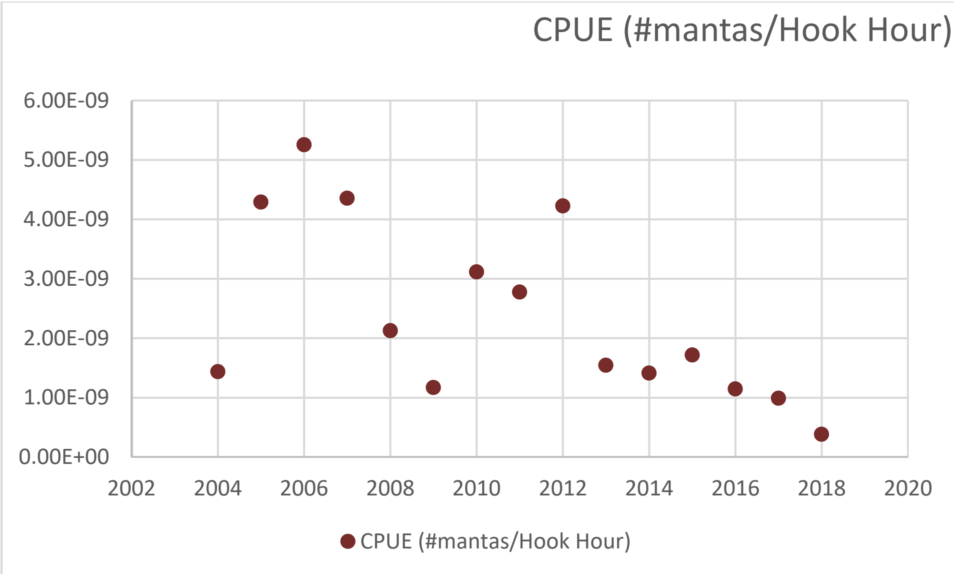

Figure 5.1. CPUE (# giant mantas/hook hour) from 2004-2018 (SEFSC data)

While the POP has a target of at least 8% observer coverage, the HMS PLL fishery has been observed at varying levels over the years and across fishing areas, often well above the 8% target. From 2005 to 2018, POP observer coverage of the fishery has varied from 7.5% to 17.9%, with an average over that period of 12.6% (SAFE 2015, SAFE 2019). We assume the same rate of interaction in the unobserved sets as observed set. Thus, with information on interactions in observed sets and observer coverage, we can calculate a rough estimate of total giant manta rays taken in the fishery from 2005-2018.

The average number of giant manta rays/year taken by observed sets of the fishery from 2005 to 2018 is 15.3 (214.2/14 years = 15.3/year). Using the average 12.6% that the fishery was observed over that time, the estimated average annual take of giant manta rays in the fishery would be 121.4 (15.3/12.6% = 121.4), which we round up to the nearest whole individual, 122 giant manta ray takes/year.

5.3.4 Estimated Giant Manta Ray Mortality in HMS PLL Gear

We now estimate the number of mortalities annually. There is limited information on immediate (dead-on-retrieval) or post-release mortality of mobula species. Beginning in 2019, the observer program is specifically gathering information related to giant manta ray interactions, such as specific identifications, hooking location, and condition on release, so we can better understand the impacts of the fishery. In the interim, data on mortality from a similar pelagic longline fishery in the Atlantic can be used as a substitute. Hooking mortality of elasmobranchs caught on the Portuguese pelagic longline fishery was analyzed by Coelho et al. (2012), where they found that mobulids (they did not differentiate by species) typically were only rarely found dead or dying when hooked or entangled. Only 1.4% of individuals were found dead. They did not estimate post-release mortality. However, it is important to note that the Portuguese fleet at the time of the study still used J-hooks almost exclusively, which results in not only a higher likelihood of hooking, but a higher likelihood of hooking in locations more likely to lead to mortality (such as hooking in the esophagus or gut), whereas since the summer of 2004, the HMS PLL fishery has been required to use circle hooks (69 FR 40733, July 6, 2004). SEFSC unpublished data from 2008 through 2016 also shows only three definitively identified giant manta rays were hooked in observed U.S. hook and line fisheries (2 in the Atlantic shark research fishery and 1 in the Gulf reef fish bottom longline fishery), and all were released alive.

Because none of the examples above are directly analogous, to be conservative and err on the side of the species, we apply the 1.4% mortality rate to the estimate of giant manta rays expected to be caught in the HMS PLL fishery. Applying the 1.4% to the 122 giant manta rays expected to be caught annually, we estimate that there will be 2 annual mortalities of giant manta rays from this fishery (122 x 1.4% = 1.7, rounded up to 2).

5.3.5 Summary of Estimated Giant Manta Rays Takes and Mortalities in the HMS PLL Fishery

In the previous section, we concluded that HMS PLL gear could take an average of 122 giant manta rays annually, resulting in 2 mortalities annually. Because this is an average annual estimate, and annual variability is expected to be high, with some years well below the average, and other years above the average, we have determined that it is appropriate to consider the potential expected take over a period of multiple years.

Thus, this opinion has determined that the HMS PLL fishery every 3 years may take up to 366 (122 x 3 = 366) giant manta rays, with 6 (2 x 3 = 6) mortalities.

5.4 Effects on Central and Southwest Atlantic DPS Scalloped Hammerhead Shark

Based on gear characteristics, scalloped hammerhead shark biology, behavior, and range, and known interactions, we believe the HMS PLL fishery may affect and is likely to adversely affect Central and Southwest Atlantic DPS scalloped sharks. This section focuses on evaluating those effects.

5.4.1 Types of Interactions and General Effects from HMS PLL Gear

Pelagic longline gear is likely to adversely affect scalloped hammerhead sharks. The impacts are primarily from hooking, but also via entanglement and trailing gear. Hooking and entanglement can lead to cuts, puncture wounds, tissue damage, and stress impacts. Hooked or entangled sharks may potentially suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns, though we have no actual evidence of such effects.

The HMS PLL fishery is required to use circle hooks. This requirement is expected to reduce the impact and severity of damage from hooking as circle hooks result in fewer hooks caught in the gut and esophagus compared to standard J-hooks. In addition, under shark bycatch mitigation measures applicable to the HMS PLL fishery, hooked or entangled sharks that are not being retained must be released using dehookers or line clippers or cutters. If using a line clipper or cutter, the shark must be released with less than three feet of line remaining attached to the hook (50 CFR 635.21(c)(6)(i)).

5.4.2 Factors Affecting the Likelihood of Central and Southwest DPS Scalloped Hammerhead Sharks Exposure to HMS PLL Gear

A variety of factors may affect the likelihood and frequency of the Central and Southwest DPS of scalloped hammerhead sharks interacting with pelagic longline gear. The spatial and temporal overlap between fishing effort and species abundance as well as species behavior may be the most evident variables involved in anticipating interactions. Other fishing related factors that may influence the likelihood and frequency of hooking and entanglement include gear characteristics and fishing techniques employed. These factors and their potential influence is discussed briefly below.

The Central and Southwest Atlantic DPS of scalloped hammerhead sharks 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. Even if scalloped hammerhead sharks 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 HMS PLL fishery 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.

Spatial/Temporal 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 PLL fishery will interact with and hook a scalloped hammerhead. The range of the Central and Southwest Atlantic DPS of scalloped hammerhead in U.S. waters falls entirely or partially within various HMS PLL fishery reporting areas (Figure 5.2). The entirety of the CAR, TUN, and TUS reporting areas occur within the DPS's range. Additionally, the southern portion of the SAR and NCA south of the 28 degrees north latitude, and almost the entirety of the FEC area, also fall into the DPS's range. 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.

Gear Usage and Fishing Techniques (Soak Times/Number of Hooks)
The amount of fishing effort affects the landing of scalloped hammerheads that are incidentally caught by the HMS PLL fishery. Number of fishers, number of trips, number of hooks, 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.

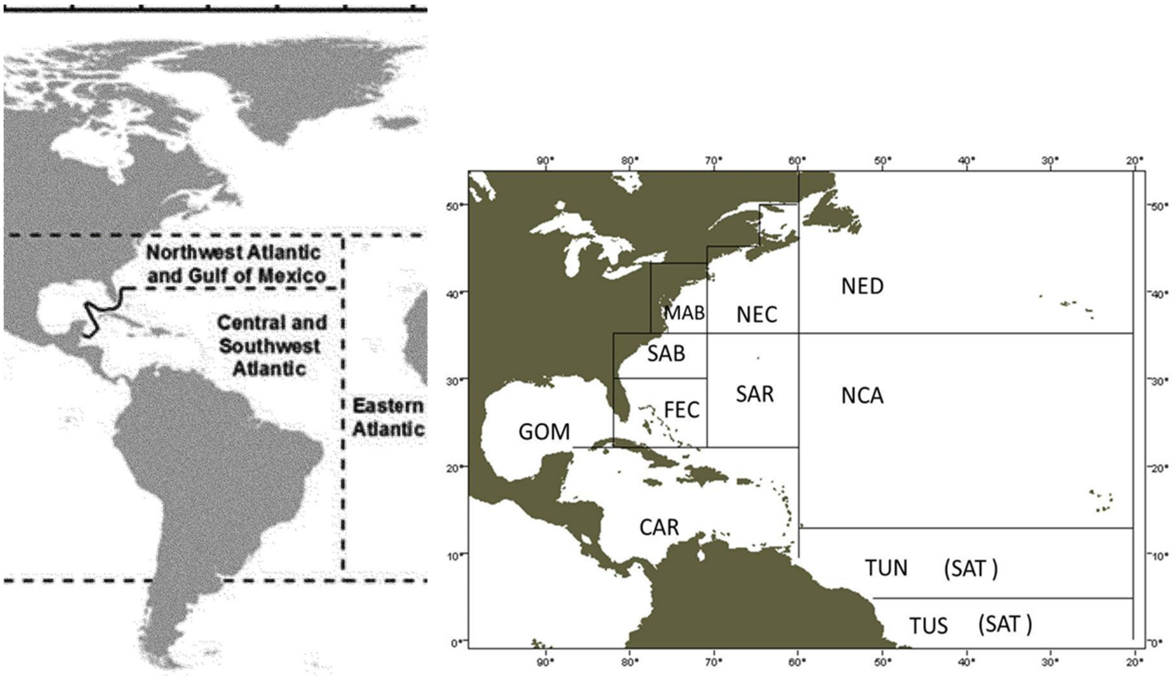

Figure 5.2 Scalloped hammerhead DPS ranges in the western Atlantic and HMS PLL fishery reporting areas (adapted from Figures 2.2 and 3.8)

5.4.3 Estimating Interactions with Central and Southwest Atlantic DPS Scalloped Hammerhead Sharks in HMS PLL gear

In order to capture the inherent variability that occurs in the fishery, we want to include enough years to do so, while ensuring that the years included are representative the fishing effort and practices expected going forward. Therefore, although we have data, and annual take estimates, all the way back to 1992, we will only use data from 2005-2018, which captures the first full year of the circle hook and bait requirements imposed on the fishery in mid-2004, under which the fishery still operates, as well as similar levels of effort.

Using observed take data and observer coverage levels from the observer program along with effort data for the fishery, the SEFSC calculated estimated takes each year in each of the HMS PLL fishery reporting areas within the boundaries of the DPS's range [SEFSC unpublished data] (see Table 5.4). The vast majority of the observed, and estimated, takes occur in the FEC reporting area. Although some small portions of the FEC lie outside of the DPS's range boundaries (primarily near Florida; see Figure 5.2), the boundaries and take locations are not precise enough to parse out differences in that area. In addition, some of those areas that are outside of the DPS's range are closed to HMS PLL fishing. We therefore take a conservative approach (erring on the side of the species) and consider all reported takes in the FEC as being within the DPS's range boundary, even though some of the takes may have been outside that boundary.

It is also important to note that in addition to the normal variability in interactions that would be expected to occur, the rarity of actual observed interactions along with the variability in observer coverage, over time and across reporting areas, results in extreme fluctuations in estimated take. One or two observed takes during a time of low observer coverage can result in a very high take estimate for that year, while a lack of observed takes due to the rarity of the event and the low observer coverage would result in an estimate of zero takes, even if takes may have occurred. While estimating rare events, such as protected species interactions, is an issue for any observed fishery, the difficulty in estimating the interactions with the Central and Southwest Atlantic DPS of scalloped hammerhead shark is exacerbated by the fact that we are looking at observer data from only a limited, defined area (the limited DPS range) as opposed to the entire range of the fishery. Consistent with the mandate to take a conservative approach and err on the side of the species, we use years that have estimated interactions (based on reported interactions) but leave out years that have no reported interactions (to avoid false "zeros" as described above). This will likely result in an overestimate of take.

Table 5.4 Central and Southwest Atlantic DPS of Scalloped Hammerhead Shark Estimated Take and Mortalities (on retrieval, not accounting for post-release mortality) in the HMS PLL Fishery 2005-2018 (SEFSC data). Note: dataset covered 2005-2018, but reported takes were rare and many years and reporting areas did not have reported takes and are not included in the table.

Year	Reporting Area	Estimated Alive on Retrieval	Coefficient of Variation (CV) for Alive	Estimated Dead on Retrieval	CV for Dead	**Estimated Total Interactions**
2008	FEC	236.2	1	258.8	0.913	495
2009	FEC	--	--	8.7	1	8.7
2010	FEC	135.2	1	--	--	135.2
	CAR	--	--	14.7	0.722	14.7
2011	FEC	529.4	0.866	45.3	0.739	574.7
2012	FEC	53.8	0.762	15.5	1	69.3
2014	FEC	--	--	38.7	0.780	38.7
2015	TUN	--	--	3.4	1	3.4
Total		**954.6**		**385.1**		**1,339.7**

Only seven of the years from 2005-2018 had observed takes, so estimates could only be derived for those years (Table 5.4). Using the method explained above, we divide the total estimated takes by seven, leaving out years in which no takes were observed. Therefore, we estimate an average of 137 individuals per year alive on retrieval (954.6/7 = 136.4 rounded up to 137) out of an estimated average of 192 total interactions per year (1,339.7/7 = 191.4 rounded up to 192).

5.4.4 Estimated Central and Southwest Atlantic DPS Scalloped Hammerhead Shark Mortality in HMS PLL Gear

In addition to estimating total interactions as detailed above, the SEFSC also estimated how many individuals were dead on retrieval, based upon POP observer data (Table 5.4). Using the same methodology described above, an estimated average of 55 individuals per year were dead on retrieval (385.1/7 = 55) annually.

The number of dead sharks on retrieval does not give us the entire picture of mortality resulting from interactions with the HMS PLL fishery. 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 must consider post-release mortality of individuals released alive, resulting from injuries sustained by the interactions.

There are currently no comprehensive studies that have determined the post-release mortality of scalloped hammerhead sharks following hooking in the HMS PLL fishery. However, a reasonable proxy can be taken from Musyl and Gilman (2019), who conducted a meta-analysis of post-release fishing mortality from a combination of various fisheries in apex predatory pelagic sharks. Although the meta-analysis included an individual study on the post-release mortality of scalloped hammerhead, we do not consider the single study to be the best available information on post fishery-interaction mortality with the HMS PLL fishery evaluated in this opinion as it was of limited size and was based on purse seine fishing. Therefore, we believe the composite meta-analysis of post-release mortality for apex predatory sharks, which includes the single study, is more representative of post-release mortality for scalloped hammerhead sharks in the HMS PLL fishery than the single study on scalloped hammerhead. Based on their meta-analysis, the authors provide multiple estimates of post-release mortality for the apex predatory sharks. We rely on the estimate that excludes what the authors have classified as "extreme" mortality events (e.g. silky sharks "braided" or wrapped up in purse seines, and thresher sharks hooked on their long caudal fins) as these "extreme" events occurred with different fishing gear (purse seine) or represented interactions typical for a different species (tail hooking on the very long tails thresher sharks use to hit and stun their prey). The remaining studies (not including the "extreme" mortalities) incorporated in the meta-analysis focus primarily on hook-and-line fisheries, including pelagic longline gear. Excluding these "extreme" mortality events, the meta-analysis concluded that there was a 20% post-release mortality for apex predatory sharks that interacted with fisheries. Therefore, we apply their general finding of 20% post-release mortality for apex predatory pelagic sharks to our scalloped hammerhead shark data. From 2005 to 2018, an average of 137 scalloped hammerhead sharks were released alive each year after interacting with the HMS PLL fishery. Based on the 20% post-release mortality, we can expect that an average of 28 (137 x 20% = 27.4 rounded to the next whole individual) sharks will suffer mortality after being released alive each year.

Therefore, the total average annual mortality of Central and Southwest Atlantic DPS scalloped hammerhead sharks resulting from interaction with the HMS PLL fishery is 83 (55 dead on retrieval + 28 post-release mortalities).

5.4.5 Summary of Estimated Central and Southwest Atlantic DPS Scalloped Hammerhead Shark Takes and Mortalities from the Proposed Action

Because of the inherently high variability in annual takes of scalloped hammerhead shark, we believe it is most appropriate to consider the take over longer periods of time instead of on an annual basis. Therefore, we consider the take on a 3-year time frame. We previously estimated 192 average annual interactions, with 55 dead on retrieval of the gear, and another 28 mortalities post-release. Expanding that to a 3-year time period, we expect 576 total interactions, with 165 mortalities on retrieval every three years. We also expect 84 post-release mortalities over that same time period, for a total of 249 mortalities every 3 years.

5.5 Effects on Oceanic Whitetip Shark

Based on gear characteristics, oceanic whitetip shark biology, behavior, and range, and known interactions, we believe the HMS PLL fishery may affect and is likely to adversely affect oceanic whitetip sharks. This section focuses on evaluating those effects.

Oceanic whitetips sharks 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. Even if oceanic whitetip sharks 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 HMS PLL fishery analyzed in this Opinion, in terms of species interactions or strikes by the vessels themselves, are not likely to adversely affect the oceanic whitetip shark.

5.5.1 Types of Interactions and General Effects from HMS PLL Gear

Pelagic longline gear is likely to adversely affect oceanic whitetip sharks. The impacts are primarily from hooking, but also via entanglement and trailing gear. Hooking and entanglement can lead to cuts, puncture wounds, tissue damage, and stress impacts. Hooked or entangled sharks may potentially suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns, though we have no actual evidence of such effects.

The HMS PLL fishery is required to use circle hooks. This requirement is expected to reduce the impact and severity of damage from hooking as using circle hooks results in fewer hooks caught in the gut and esophagus compared to standard J-hooks. In addition, under shark bycatch mitigation measures applicable to the HMS PLL fishery, hooked or entangled sharks that are not being retained must be released using dehookers or line clippers or cutters. If using a line clipper or cutter, the shark must be released with less than three feet of line remaining attached to the hook (50 CFR 635.21(c)(6)(i)).

5.5.2 Factors Affecting the Likelihood of Oceanic Whitetip Shark Exposure to HMS PLL Gear

A variety of factors may affect the likelihood and frequency of the oceanic whitetip sharks interacting with pelagic longline gear. The spatial and temporal overlap between fishing effort and species abundance as well as species behavior may be the most evident variables involved in anticipating interactions. Other fishing related factors that may influence the likelihood and frequency of hooking and entanglement include gear characteristics and fishing techniques employed. These factors and their potential influence is discussed briefly below.

Spatial/Temporal 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.

Gear Usage and Fishing Techniques (Soak Times and Number of Hooks)
The amount of fishing effort affects the likelihood of oceanic whitetip sharks being incidentally caught on the HMS PLL gear. Number of fishers, number of trips, number of hooks, and length of time the gear is left in the water (soak time) are all important considerations. More fishing likely increases the probability of hooking this species.

5.5.3 Estimating Interactions with Oceanic Whitetip Sharks in HMS PLL Gear

In order to capture the inherent variability in interactions that occurs in the fishery, we want to include enough years to do so, while ensuring that the years included are representative of the fishing effort and practices expected going forward. Therefore, although we have data, and annual take estimates, all the way back to 1992, we will only use data from 2005-2018, which captures the first full year of the new circle hook and bait requirements imposed on the fishery in 2005, under which the fishery still operates, as well as similar levels of effort.

Using observed take data from the observer program along with effort data for the fishery, the SEFSC calculated estimated takes each year [SEFSC unpublished data] (see Table 5.5). Annual estimated total interactions from 2005-2018 varied greatly, with a low of 186.7 and a high of 1042.9. The average annual number of interactions over that time period was estimated to be 453.9, so rounding up to the nearest whole individual, we estimate that on average we can expect 454 interactions per year with oceanic whitetip sharks by the HMS PLL fishery.

Table 5.5 Oceanic Whitetip Shark Estimated Take and Mortalities (on retrieval, not accounting for post-release mortality) in the HMS PLL Fishery 2005-2018 (SEFSC data)

(Note that 2014 and 2016 had estimates for unknown disposition, as follows: 2014: 2.4282; 2017: 2.5279. To be conservative, those were added in as “dead” in the table below. The CVs for 2014 and 2017 dead are based on numbers before adding in the estimates with unknown dispositions.)

Year	Estimated Alive on Retrieval	CV for Alive	Estimated Dead on Retrieval	CV for Dead	**Estimated Total Interactions**
2005	484.0	0.228	155.0	0.368	**639.0**
2006	229.6	0.265	29.6	0.580	**259.2**
2007	133.1	0.236	53.6	0.434	**186.7**
2008	283.3	0.203	21.0	0.533	**304.3**
2009	297.8	0.187	121.8	0.271	**419.6**
2010	234.8	0.338	60.3	0.402	**295.1**
2011	178.4	0.293	63.5	0.517	**241.9**
2012	364.8	0.206	70.8	0.391	**435.6**
2013	449.9	0.244	87.9	0.448	**537.8**
2014	384.6	0.192	120.9	0.378	**505.5**
2015	332.4	0.193	113.4	0.335	**445.8**
2016	426.6	0.151	129.2	0.243	**555.8**
2017	871.4	0.125	171.4	0.444	**1042.8**
2018	382.8	0.164	101.6	0.284	**484.4**
Average	**361.0**		**92.9**		**453.9**

5.5.4 Estimated Oceanic Whitetip Shark Mortality in HMS PLL Gear

In addition to estimating total interactions as detailed above, the SEFSC also estimated how many individuals were dead on retrieval, based upon observer data, extrapolated to the entire fishery based on effort data [SEFSC unpublished data] (Table 5.5). Based upon those calculations, an estimated average of 93 (rounded up from 92.9) oceanic whitetip sharks were retrieved dead every year. Note that number is slightly increased by the inclusion of an estimated 2.4 (from 2014) and 2.5 (from 2017) sharks of unknown disposition that we have included as “dead” as part of our approach of making conservative assumptions in favor of the species.

The number of dead sharks on retrieval does not give us the entire picture of mortality resulting from interactions with the HMS PLL fishery. 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 must consider post-release mortality of individuals released alive, resulting from injuries sustained by the interactions.

There are currently no comprehensive studies that have determined the post-release mortality of oceanic whitetip sharks following hooking in the HMS PLL fishery. However, a reasonable proxy can be taken from Musyl and Gilman (2019), who conducted a meta-analysis of post-release fishing mortality from a combination of various fisheries in apex predatory pelagic sharks. Although the meta-analysis included two individual longline studies on oceanic whitetip sharks, and Musyl and Gilman (2019) concluded a weighted summary effect of 11% post-release mortality for oceanic whitetip shark from those studies, we do not consider it to be the best available information on post fishery-interaction mortality with the HMS PLL fishery as they were of very limited size (15 tagged sharks). Rather than relying on these studies alone, we look to the meta-analysis, which compiles information from 27 studies and 346 tagged sharks, including the information from the oceanic whitetip studies (15 tagged oceanic whitetips). Based on their meta-analysis, the authors provide multiple estimates of post-release mortality for the apex predatory sharks. We utilize the author's meta-analysis conclusion for apex predatory pelagic sharks that excluded what they deemed "extreme" mortality events (e.g. silky sharks "braided" or wrapped up in purse seines, and thresher sharks hooked on their long caudal fins). We believe that analysis is more appropriate than the analysis including the "extreme" mortality events because the extreme events occurred with different fishing gear (purse seine) or represented interactions typical for a different species (tail hooking on the very long and vulnerable tails thresher sharks use to hit and stun their prey). The remaining studies incorporated in the meta-analysis focus primarily on hook-and-line fisheries, including pelagic longline gear, and therefore we believe it is more representative than the broader analysis with the "extreme" mortalities. Excluding these "extreme" mortality events, the meta-analysis concluded that there was a 20% post-release mortality for apex predatory sharks that interacted with fisheries. Therefore, we apply their general finding of 20% post-release mortality for apex predatory pelagic sharks to our oceanic whitetip shark data. As shown in Table 5.5, from 2005 to 2018, an average of 361 oceanic whitetip sharks were released alive each year after interacting with the HMS PLL fishery. Based on the 20% post-release mortality, we can expect that an average of 73 (361 x 20% = 72.2 rounded to the next whole individual) sharks will suffer mortality after being released alive each year.

Therefore, the total average annual mortality of oceanic whitetip sharks resulting from interaction with the HMS PLL fishery is 166 (93 dead on retrieval + 73 post-release mortalities).

5.5.5 Summary of Estimated Oceanic Whitetip Shark Takes and Mortalities from the Proposed Action

Because of the inherently high variability in annual takes of oceanic whitetip shark, we feel it is most appropriate to consider the take over longer periods of time instead of on an annual basis. Therefore, we consider the take on a 3-year time frame. We previously estimated 454 average annual interactions, with 93 dead on retrieval of the gear, and another 73 mortalities post-release. Expanding that to a 3-year time period, we expect 1,362 total interactions, with 279 mortalities on retrieval every three years. We also expect 219 post-release mortalities over that same time period, for a total of 498 mortalities every 3 years.

6.0 Cumulative Effects

Cumulative effects include the effects of future state or private activities, not involving federal activities, that are reasonably certain to occur within the action area of the federal action considered in this Opinion (50 CFR 402.02). 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. Cumulative effects from unrelated, non-federal actions occurring in the action area may affect sperm whales, sea turtles, giant manta rays, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, and oceanic whitetip sharks.

The activities discussed in Sections 3 and 4 of this Opinion described as occurring within the action area are expected to continue as described into the foreseeable future, concurrent with the proposed action. The primary actions or effects from human activities include vessel operations, pollution/marine debris, and climate change.

Watercraft are the greatest contributors to overall noise in the sea and have the potential to interact with protected species though vessel strikes. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on sperm whales, sea turtles, and giant manta rays may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. Vessel interactions with pelagic sharks such as scalloped hammerhead and oceanic whitetip sharks is far less likely because of limited time at the surface. Commercial traffic and recreational pursuits can also adversely affect sea turtles, giant manta rays, and sperm whales through vessel strikes (hull impacts and/or propellers). 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.

Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them to species in this Opinion, however, the level of impacts cannot be projected. Marine debris (e.g., debris, discarded fishing line or lines from boats) can entangle animals, resulting in mortality, or impair their normal movement and behavior. For example, sea turtles, sharks, rays, and marine mammals have been documented stranded in the U.S. entangled in plastics, monofilament, discarded netting, and many other waste items. Entanglement can lead to death, injury, mutilation, starvation, and increased susceptibility to predation. Ingestion of plastic, rubber, fishing line and hooks, tar, string, Styrofoam, epoxy, and aluminum has been documented in marine species, potentially resulting in digestive tract impaction or toxic absorption.

Global climate change is likely adversely affecting sea turtles, whales, sharks, and rays. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The effects on ESA-listed species are unknown at this time. There are multiple hypothesized effects to 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 animals, including earlier onset of nesting, shorter intervals between nesting, and a decrease in the length of nesting season for sea turtles. Water temperature is a main factor affecting the distribution of large whales, and may affect the range of these species. A decline in reproductive fitness as a result of global climate change could have effects on the abundance and distribution of animals in the Atlantic Ocean, including the Gulf of Mexico and Caribbean Sea.

NMFS is not aware of any proposed or anticipated changes in these factors that would substantially change the impacts each has on ESA-listed sea turtles, sharks and rays, and sperm whales covered by this Opinion. Therefore, NMFS expects that the levels of effects described for each of the factors will continue at similar levels into the foreseeable future.

7.0 Jeopardy Analysis

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 sperm whales, sea turtles, giant manta rays, scalloped hammerhead sharks, or oceanic whitetip sharks. 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 one 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. To do so, we first look at the impact at the level of the population or populations that are expected to be affected by the proposed action. If the proposed action is not expected to appreciably reduce the likelihood of survival and recovery of the affected populations, we would likewise conclude that the it is not expected to appreciably reduce the likelihood of survival and recovery of the globally-listed species as a whole. If we determine that the proposed action is expected to appreciably reduce the likelihood of survival and recovery for the affected populations, we would then further analyze the impacts to determine whether that impact to the affected population is expected to appreciably reduce the likelihood of survival and recovery for the globally-listed species as a whole. 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 Sperm Whales

As discussed in the *Status of Species,* sperm whales are listed as endangered under the ESA. Sperm whales occur in all oceans of the world. The best estimate of the current worldwide abundance of sperm whales is between 300,000 and 450,000 individuals (Whitehead 2002). Within the Atlantic, their abundance is estimated at 90,000 to 134,000 individuals and within the Gulf of Mexico, there are between 763 (NMFS 2015c) and 2,128 (Roberts et al. 2016a) resident whales, with the best estimate of 1,436 (Hayes et al. 2020, in review). Both of the observed takes of sperm whales in the HMS PLL since 1992 have occurred in the Gulf of Mexico. While there are no long-term estimates of abundance trends within the Gulf of Mexico, sperm whales in this region are thought to have been heavily impacted by the DWH oil spill, which may have resulted in a population decline (Chiquet et al. 2013). That said, sperm whales are likely one of the most abundant large whale species, and on a global scale, they show little genetic differentiation in terms of nDNA, meaning there is genetic mixing and conductivity across the species on a global scale, likely due to male-mediated gene flow from male sperm whales roaming widely. However, within ocean basins, and even more so within semi-enclosed basins such as the Gulf of Mexico, sperm whales do show some genetic differentiation based on mtDNA, which is thought to be the consequence of shorter-ranging, in some cases resident, females. So while there is mixing globally throughout the species, there are regional and basin-wide stocks that have some degree of genetic differentiation from each other resulting from short-ranging or resident females and their offspring staying in the same basin or region for generations. As none of the stocks for which data are available have high levels of genetic diversity, the species may be at some risk to inbreeding and 'Allee' effects (low population growth rates due to low density and limited mate availability), although the extent of this risk is currently unknown.

The Status of Species and Environmental Baseline Sections indicate the primary reason for sperm whale ESA-listed status is historical commercial whaling. With the threat of large-scale commercial whaling now gone, sperm whales have shown strong signs of recovery with higher estimates of their abundance perhaps approaching population sizes prior to commercial whaling. That said, they still face several threats including vessel interactions, incidental capture in fishing gear, habitat degradation (including pollution and sound), and military operations. In addition, as noted above, sperm whales in the Gulf of Mexico were likely impacted by the 2010 DWH oil spill. Cumulative Effects expected to affect the sperm whales in the future include effects from activities similar to those identified in the Status of Species and Environmental Baseline, which are expected to continue in the future at similar levels.

We conservatively estimated that the proposed action would result in the take (entanglement in HMS PLL gear) of 3 sperm whales every 10 years in the Gulf of Mexico, in any combination of lethal and non-lethal. We do not expect the non-lethal take associated with the proposed action to reduce the numbers or reproduction of the species. The non-lethal take also will not reduce the distribution of the species as individuals caught alive will be released in the same area the interaction occurred.

The lethal take of the species is a reduction in numbers and could reduce the species' reproduction because the lethal take could involve a female, thus removing her reproductive potential. We analyze the effect to the sperm whales using the conservative assumption that all 3 takes over each 10-year period will be lethal. However, we do not expect the lethal interaction to affect the species' distribution because 3 individuals lost every 10 years represents a very small percent of the population in the Gulf of Mexico, and would not alter the presence of sperm whales in that basin. Whether the reduction in numbers and reproduction will appreciably reduce the species' likelihood of survival in the wild depends on the species' response to these reductions. The loss of 3 individuals every 10 years represents 0.14-0.39 percent of the estimated 763-2,128 sperm whales in the Gulf of Mexico. The number of sperm whales in the Gulf of Mexico represents at a maximum 2 percent of all sperm whales in the Atlantic, and less than one percent of the species abundance globally (sperm whales are globally listed under the ESA). Given this small percentage, we find that the proposed action is not likely to jeopardize the continued existence of the sperm whale by appreciably reducing the likelihood of the survival in the wild.

The Recovery Plan for sperm whales (NMFS 2010) states the recovery goal is to "promote recovery of sperm whales to a point at which they can be downlisted from endangered to threatened status, and ultimately to remove them from the list of Endangered and Threatened Wildlife and Plants, under the provisions of the ESA." The Recovery Plan includes the following objectives that are relevant to the proposed action:

> *Objective 1*: Achieve sufficient and viable populations in all ocean basins.
>
> *Objective 2*: Ensure significant threats are addressed.

We believe that the estimated mortality of up to 3 sperm whales every 10 years is not counter to Objective 1, and will not prevent the species from attaining sufficient and viable populations in the Atlantic and Gulf of Mexico. As detailed above, that loss represents only 0.14-0.39% of just the Gulf of Mexico population, assuming that interactions and mortalities will occur in the Gulf of Mexico, and the Gulf of Mexico population is at most about 2 percent of the total Atlantic population. Likewise, we believe that the proposed action is not contrary to Objective 2, as the information provided in the above analysis demonstrates that the fishery is not a significant threat to the species.

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 sperm whales in the wild, and therefore the proposed action is not likely to jeopardize the continued existence of the species.

7.2 Sea Turtles

Some sea turtle species are listed as a single species distributed globally (leatherback, Kemp's ridley, hawksbill, and olive ridley); therefore, a jeopardy determination must evaluate whether 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 evaluate whether the proposed action will appreciably reduce the likelihood of survival and recovery of these DPSs in the wild.

7.2.1 Loggerhead Sea Turtles (NWA DPS)

The proposed action may result in up to 1,080 loggerhead sea turtle takes, 800 of which are expected to be nonlethal and 280 of which are expected to be lethal, every 3 years. The potential nonlethal capture and release of 800 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 experiencing nonlethal injuries are expected to fully recover such that no reductions in reproduction or numbers of loggerhead sea turtles are anticipated. 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 280 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. Adult and sub-adult juveniles primarily reside in coastal waters, while the individuals residing in the open ocean areas where the HMS PLL fishery operates would likely be pelagic stage juveniles that can be 10 or more years from reproductive age. The HMS PLL fishery is expected to interact primarily with these pelagic stage juveniles, which are making the transition between pelagic and benthic modes and are not yet of breeding age. Pelagic juveniles will encounter many other potential sources of mortality, natural and anthropogenic, prior to reaching maturity. However, some of these individuals would be females that would have survived the other threats and reproduced in the future, thus eliminating those 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. Therefore, the loss of pelagic juvenile females that would have survived to adulthood 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 throughout the proposed action area, 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, 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. 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 too 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. These numbers were derived prior to additional years of increased nesting.

Florida accounts for more than 90% of U.S. loggerhead nesting. 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 non-significant 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 from 2012-2016 resulting in widening confidence intervals. 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. However, it is important to note that with the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.), it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani, et al. 2019).

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 level or increasing over the years. Additionally, our estimate of future takes is not a new source of impacts on the species; effort in the fishery is expected to stay largely the same, and since 2005, which is the first full year when the revised gear regulations were implemented, the impacts of the fishery have been reduced relative to decades prior. Additionally, the total expected take and total mortality are lower than the expected take calculated in the 2004 Opinion (280 mortalities (this Opinion) vs. 339 (2004 Opinion) every 3 years), which resulted in a non-jeopardy determination.

The proposed action could remove up to 280 individuals every 3 years. These removed individuals represent approximately 0.073% every 3 years of the low end of the NMFS-SEFSC (2011) estimate of 381,941 loggerheads within the NW Atlantic continental shelf (as opposed to pelagic juveniles on the open ocean). However, because many of the pelagic juveniles suffering mortality because of the fishery would not otherwise have survived to adulthood (and would not have become part of the more coastal population), the percentage reduction of the coastal shelf population as a result of lethal take in this fishery is likely less than the 0.073%. In addition as we noted above, this estimate reflects a subset of the entire loggerhead population in the western North Atlantic Ocean, and thus these individuals represent an even smaller proportion of the population removed. The number of pelagic juveniles is unknown, but may exceed that of coastal individuals. While the loss of 280 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 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 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 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 stable or increasing over the past couple of decades. 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 280 loggerhead sea turtles over the future every 3 years is so small in relation to the overall population on the continental shelf (which does not include the large, but unknown pelagic population numbers), that it would be hardly detectable. For these reasons, we do not believe the proposed action will impede achieving this recovery objective.

The proposed action is not counter to the recovery plan's Objective Nos. 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 of pelagic juveniles may still occur during the proposed action, 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.

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, and therefore the proposed action is not likely to jeopardize the continued existence of the species.

7.2.2 Leatherback Sea Turtles

The proposed action may result in up to 996 leatherback sea turtle takes, 275 of which are expected to be lethal, every 3 years. The nonlethal capture of 721 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 275 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. Unlike hardshell sea turtle species, leatherbacks are pelagic throughout their lives and primarily occur in the open ocean. Therefore, interactions with the fishery include both adults and larger juveniles. Lethal captures could also result in a potential reduction in future reproduction, assuming some of these individuals would be female and are either adults or juveniles that would have survived otherwise to reproduce in the future. 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. As detailed in the Status of the Species section above, an individual adult female can have a fertility span up to 25 years (Hughes 1996), nesting on average every 2-4 years during that time span (Garcia M. and Sarti 2000; McDonald and Dutton 1996; Spotila et al. 2000). When nesting, females lay up to 10 nests with 100 or more eggs per nest (Eckert et al. 2012; Eckert 1989; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). Thus, 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. 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 Section 3, 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. The effects of the HMS PLL fishery have been occurring for years, and since 2005, which is the first full year when the revised gear regulations were implemented, the impacts of the fishery have been reduced relative to the decades prior. 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 275 leatherback sea turtles every 3 years accounts for only a small fraction of the population (0.3 to 0.7% if all takes were from the adult population, which is an overestimate as juveniles occur in the same areas, interact with the fishery, and are greater in number than adults) of the North Atlantic population estimates, which is a subset of the listed entity. While we do not have more updated population estimates, and later trend analyses on the nesting beaches now show declines (see Section 3, Status of the Species), this information serves to give a sense of the scope of the population-level impact. We do not believe these potential losses will have a detectable impact on the population numbers.

Until recently, of the 15 leatherback nesting populations in the Northwest Atlantic, 7 showed an increase in nesting (Florida, Puerto Rico [not Culebra], St. Croix-U.S. Virgin Islands, British Virgin Islands, Trinidad, Guyana, and Brazil) and 3 were showing 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) appeared to have remained stable during that time (2009-2015). 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). However, the Northwest Atlantic Leatherback Working Group (2018) has determined that there is an ongoing decline in the nesting trend of the Southern Caribbean/Guianas stock, which includes French Guiana and Suriname, in the short term (10.43% annual geometric mean decline from 2008-2017) that is also driving a longer-term decline in the trend (5% annual geometric mean decline from 1990-2017) for this stock. Other stocks within the Northwest Atlantic have also contributed to the regional decline, with the Western Caribbean stock showing a nesting decline of almost 6%, the Northern Caribbean stock with a decline of 10%, and the Florida stock with a decline of almost 7% annual geometric mean from 2008-2017, though the long-term nesting trend (1990-2017) in Florida appears to be increasing (Northwest Atlantic Leatherback Working Group 2018). The working group has identified a number of possible drivers for the recent declines, including habitat loss, other anthropogenic impacts (including fisheries that may have increased in scope and/or developed near nesting beaches), and possible changes in demographic and life history factors. However, it is not clear which factor is driving the declines. The HMS PLL fishery has been affecting this population for decades, with a reduction in impacts starting in 2005 after the sea turtle conservation gear requirements were fully in place.

As described above, and in more detail in Section 3, Status of the Species, aside from the long-term nesting trend in Florida, most all of the other nesting populations appear to be decreasing, reversing the stable and increasing trend that was observed earlier, as of data through 2017. However, since we anticipate 275 mortalities (both large juveniles and adults) every 3 years, a fraction of the reduced but still large overall nesting population, and we have no reason to believe nesting females will be disproportionately affected, we believe the potential mortality 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 discussed above, an updated analysis from 2018 has shown a reverse in trends, as the Culebra, St. Croix, and Florida nesting populations have decreased in recent years, although the long-term trend in Florida remains positive. It is unclear whether the declines represent a shift in nesting locations, changes in reproductive output, actual declines in the adult female population, or some combination of those factors. Since we concluded that the potential loss of up to 275 leatherback sea turtles every 3 years is not likely to have any detectable effect on the overall nesting trends in the Northwest Atlantic, and there is no basis to believe the fishery impacts individuals from these particular nesting beaches at a different rate as individuals from other nesting beaches, 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.2.3 Kemp's Ridley Sea Turtle

The proposed action may result in up to 21 "other hardshell" sea turtles takes every 3 years, of which 8 are expected to be lethal and 13 are expected to be non-lethal. These "other hardshell" sea turtles can be any combination of the non-loggerhead hard shell sea turtles, i.e., Kemp's ridley, green, hawksbill, or olive ridley sea turtles. Because of the limited data, we are not able to identify the specific number of each of these species that will interact with the fishery. However, interactions with each of these species is possible given historical interactions. Because we are not able to identify take numbers at the species level, for the purpose of analyzing whether the proposed action is likely to jeopardize the continued existence of any of the "other hardshell" sea turtles, we will assume that all 21 of the takes could be of each of the species (e.g., all 21 could be Kemp's ridley, all 21 could be green sea turtles). This allows us to estimate the worst case scenario for each of these species, erring on the side of caution for each species.

Thus, for the purpose of evaluating the likelihood that the proposed action will jeopardize the continued existence of Kemp's ridley sea turtles, we assume that the proposed action may result in up to 21 Kemp's ridley sea turtle takes, of which 8 are expected to be lethal and 13 are expected to be nonlethal, every 3 years. The nonlethal capture of 13 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 8 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 8 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), then declines in 2018 and 2019, when only 11,090 nests were recorded (Gladys Porter Zoo nesting database 2019). 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; 2018-2019), 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 turtle 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, we have seen positive trends in the status of this species, despite the ongoing operation of the HMS PLL fishery. 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 8 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 2019, indicate there were a total of 11,090 nests on the main nesting beaches in Mexico. This number represents approximately 4,436 females nesting that 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 and 2019. Although we believe there is a long-term increasing trend in nesting that is evidence of an increasing population, 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 8 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.2.4 Green Sea Turtles (North Atlantic and South Atlantic DPS)

The proposed action may result in up to 21 "other hardshell" sea turtles takes every 3 years, of which 8 are expected to be lethal and 13 are expected to be non-lethal. These "other hardshell" sea turtles can be any combination of the non-loggerhead hardshell sea turtles. However, as explained above, because of data limitations, we do not know which of the species will be taken in which amounts. Therefore, for the purpose of evaluating the likelihood of whether the proposed action will jeopardize the continued existence of green sea turtles, we assume that the proposed action may result in up to 21 takes of green sea turtles, of which 8 are expected to be lethal, every 3 years.

Takes of green sea turtles could be of individuals from either the North Atlantic or South 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 North 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, and assume only 5% of the turtles that interact with the fishery come from the South Atlantic DPS, relying on the estimate from the Atlantic coast Florida study given the fishery's broad operation throughout the Atlantic and Gulf of Mexico. Based on the total anticipated take of up to 21 green turtles, including 8 mortalities, we applied the estimate of 5% of the individuals being from the South Atlantic DPS as presented above. Therefore, we estimate that 20 of the green turtles taken would be from the North Atlantic DPS (21 x 0.95 = 19.95, rounded up to the nearest whole turtle), with 8 mortalities (8 x 0.95 = 7.6, rounded up to the nearest whole turtle). We estimate that 2 of the individuals taken by the fishery would be from the South Atlantic DPS (21 x 0.05 = 1.05, rounded up to the nearest whole turtle), with 1 mortality (8 x 0.05 = 0.4, rounded up to the nearest whole turtle).

7.2.4.1 North Atlantic DPS

The proposed action may result in 20 green sea turtle takes from the North Atlantic DPS (12 nonlethal, 8 lethal) every 3 years. The potential nonlethal capture of 12 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 8 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, 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 8 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 1989-2019, 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 almost 41,000 in 2019 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). 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 8 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 given the estimated population size and increasing trend. 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.2.4.2 South Atlantic DPS

As described above, for the purpose of analyzing whether the proposed action is likely to jeopardize the continued existence of green sea turtles, we assumed all 21 takes of "other hardshell" sea turtles (8 lethal, 13 non-lethal) expected every 3 years were green sea turtles, and of those, the proposed action may result in up to 2 green sea turtle captures from the South Atlantic DPS (1 nonlethal, 1 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 1 green sea turtle 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, 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 (Catry 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 1 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 1 South Atlantic DPS green sea turtle every 3 years will result in a reduction in numbers when capture occurs, but it is unlikely to have any detectable influence on the trends noted above. Nonlethal capture of a sea turtle 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.2.5 Hawksbill Sea Turtles

The proposed action may result in up to 21 "other hardshell" sea turtles takes every 3 years, of which 8 are expected to be lethal and 13 are expected to be non-lethal. These "other hardshell" sea turtles can be any combination of the non-loggerhead hardshell sea turtles, i.e., Kemp's ridley, green, hawksbill, or olive ridley sea turtles. As described above, because of data limitations, for the purpose of analyzing whether the proposed action is likely to jeopardize the continued existence of any of the "other hardshell" sea turtles, we will assume that all 21 of the takes could be of each of the species (e.g., all 21 could be hawksbill sea turtles).

Thus, for the purpose of evaluating whether the proposed action is likely to jeopardize the continued existence of hawksbill sea turtles, we assume that the proposed action may result in up to 21 takes of hawksbill sea turtles, of which 8 are expected to be lethal and 13 are expected to be non-lethal, every 3 years. The nonlethal capture of 13 hawksbill 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 hawksbill sea turtles is anticipated.

The lethal take of up to 8 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 8 hawksbills every 3 years would equal only 0.036% 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).

We have still seen positive trends in the status of this species with the operation of the fishery. 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 8 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 Antiqua/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. 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 8 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 past interactions, and even with operation of the fishery, 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. Although 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 8 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.2.6 Olive Ridley Sea Turtles

The proposed action may result in up to 21 "other hardshell" sea turtles takes every 3 years, of which 8 are expected to be lethal and 13 are expected to be non-lethal. These "other hardshell" sea turtles can be any combination of the non-loggerhead hardshell sea turtles. However, as explained above, because of data limitations, we do not know which of the species will be taken in which amounts. Therefore, for the purpose of evaluating the likelihood of whether the proposed action will jeopardize the continued existence of olive ridley sea turtles, we assume that the proposed action may result in up to 21 takes of olive ridley sea turtles, of which 8 are expected to be lethal, every 3 years.

There is no Recovery Plan for the Atlantic olive ridley turtle because they are viewed as being largely outside of U.S. jurisdiction. The HMS PLL fishery is one of the few U.S. activities that operate in the range of the Atlantic olive ridley, in the oceanic waters off the Atlantic coast of northern South America. Recorded interactions with Atlantic olive ridleys in this fishery have been rare and sporadic, with 4 documented individuals over the past 14 years (2005-2018), all occurring in the TUN fishery reporting area. However, as explained above, because of data limitations and unidentified turtles, we assume that all of the 21 takes (8 lethal) of "other hardshell" sea turtles every three years could be olive ridley sea turtles, providing for a highly conservative analysis on the potential impact to the species. The nonlethal capture of 13 olive 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 this species are anticipated. Since the species would be released within the general area where caught, no change in the distribution of olive ridley sea turtles is anticipated. The 8 lethal interactions every 3 years represents a reduction in numbers and potential reduction in reproduction, assuming the individuals are female and are or would have survived to reproductive age. Given that olive ridley sea turtles generally have large ranges, no reduction in the distribution of olive ridley sea turtles is expected from the proposed action, which only has the potential to interact with the species in a narrow portion of its range.

Overall, there is limited information on the status and trends for olive ridleys in the Atlantic, where the impacts of the proposed action are expected to occur. We can deduce, based on information presented above about olive ridley life history and distribution that sea turtles found within the range of the HMS PLL fishery off South America would most likely come from the western Atlantic breeding populations, and not the eastern Atlantic population. We know that since its listing under the ESA, there has been an overall decline in abundance of this species in the western Atlantic, probably the result of continued direct and incidental take, particularly in shrimp trawl nets and nearshore gill nets. The western North Atlantic (Suriname, French Guiana, and Guyana) nesting population has declined more than 80 percent since 1967. However, as noted above nesting in Suriname and French Guiana may be showing signs of stabilizing, and the very small nesting population in Brazil may be increasing. Given the small number of mortalities (8 every 3 years) estimated, even using a highly conservative analysis, and the potentially stabilizing western Atlantic population, we do not believe that the proposed action is likely to affect these trends and reduce appreciably the likelihood of the species survival. Because there is no recovery plan or other recovery guidance for the Atlantic olive ridley, we do not have recovery goals and priorities from which to assess the impacts, as we do for other species. However, recovery is the process by which the ecosystems of olive ridley 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). Thus, the first step in recovering a species is to reduce identified threats; only by alleviating threats can lasting recovery be achieved. Threats to the species include harvest of eggs and killing of adults on nesting beaches, bycatch in fishing gear, vessel strikes, and ocean pollution and marine debris. The proposed action will not impede the process of restoring the ecosystems that affect olive ridley sea turtles nor is the HMS PLL fishery a significant threat. The reduction in numbers and future reproductive potential, as described above, are very small (lethal take of 8 individuals every 3 years) even using a very conservative analysis, and would not be expected to appreciably reduce the species' likelihood of recovery given the scope of the effect relative to the species' overall range, population, and reproductive potential.

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 olive ridley sea turtle in the wild. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.3 Giant Manta Rays

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 366 total giant manta ray takes over consecutive 3-year periods, with up to 6 mortalities in the same time frame. The nonlethal capture of 360 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 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 lethal take of 6 giant manta rays over consecutive 3-year periods will 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 lethal take is also expected to result in the loss of reproduction. However, given that in the Atlantic the species is not thought to be in peril as it is elsewhere in its range, we believe the lethal take of 6 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, 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). Thus, 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, the 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. The proposed action will not impede the process of restoring the ecosystems that affect giant manta rays nor are these fisheries a significant threat (366 average takes/3-years with 6 mortalities).

While there is no recovery plan available yet, NMFS has developed a Giant Manta Ray Recovery Outline (https://www.fisheries.noaa.gov/resource/document/giant-manta-ray-recovery-outline). The recovery outline "is meant to serve as an interim guidance document to direct recovery efforts, including recovery planning, for the giant manta ray until a full recovery plan is developed and approved." It presents a preliminary strategy for recovery of the species, as well as recommended high priority actions to stabilize and recover the species.

The interim recovery strategy focuses on (1) stabilizing population trends through a reduction of threats, so the species is no longer declining and (2) gathering additional information on the species' current distribution and abundance, movement and habitat use of adult and juveniles, mortality rates in commercial fisheries (including at-vessel and post-release mortality), and other potential threats that may contribute to the species' decline. As discussed above, NMFS does not believe that U.S. fisheries are contributing significantly to the identified threats. Therefore, the proposed action is not expected to impede the stabilization of population trends for the species. Additionally, the inclusion of giant manta ray information in observer reports for the HMS PLL fishery will contribute to the second goal of gathering information on the species.

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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

7.4 Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS

The Central and Southwest Atlantic DPS of scalloped hammerhead shark occurs within a portion of 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 576 (327 non-lethal and 249 lethal) scalloped hammerhead shark takes over consecutive 3-year periods. The nonlethal capture of 327 scalloped hammerhead sharks every 3 years (average 109 per year), 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 would occur in the boundaries of the DPS's range, and individuals would be released within the same general area where caught, no change in the distribution of this species is anticipated.

We estimate that up to 249 of those takes over three years may be lethal (average of 83 annually). The loss of 249 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 reproductive value as compared to the reproductive value in the absence of the proposed action, if a female is taken. While we have no reason to believe the proposed action will disproportionately affect females or adults, the loss of an adult female could preclude the production of future progeny. 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 249 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 249 scalloped hammerhead sharks every 3 years (83 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 an annual loss of 83 scalloped hammerheads will not significantly decrease the populations within the Central and Southwest Atlantic DPS as this is a limited amount of loss relative to the population size, nor will it change their distribution. It is important to note that the subject fishery is not a new impact to the species in those areas, and has been ongoing since before the late 1990s. Later changes to the fishery (2004 requirement to use circle hooks for sea turtle conservation) may also have ancillary benefits for sharks via the possible reduction of gut and esophageal hooking. Additionally, on August 29, 2011, NMFS finalized a rule domestically implementing ICCAT Recommendation 10-08, which prohibits the retention, transshipping, landing, sorting, or selling of hammerhead sharks caught in the HMS PLL fishery (76 FR 53652; August 29, 2011). Therefore, it can be reasonably assumed that prior to 2011 at least some individuals were retained in the HMS PLL fishery, whereas now all scalloped hammerheads must be released with no more than 3 feet of trailing line. 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 (576 total 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

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 1,362 oceanic whitetip shark takes over consecutive 3-year periods. We estimate that a total of 498 of those interactions may be lethal, including both dead on retrieval and post-release mortalities, leaving 864 non-lethal takes every 3 years.

The nonlethal capture of 864 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 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.

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). It is important to note that the subject fishery is not a new impact to the species in those areas, and has been ongoing since before the late 1990s.

The loss of 498 oceanic whitetip sharks over consecutive 3-year periods could result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed action, if females are taken. While we have no reason to believe the proposed action will disproportionately affect females or adults, the loss of an adult female oceanic whitetip shark could preclude the production of future progeny. 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.

Likewise, the loss of those individuals would represent a reduction in numbers 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. However, we believe that the loss in numbers and reproduction are likely small relative to the species size and reproductive potential. Additionally, the populations within the action area considered in this Opinion are thought to have stabilized since 2000 or earlier, during which time the fishery, and its impacts, was already occurring. Later changes to the fishery (2004 requirement of circle hooks for sea turtle conservation) may also have ancillary benefits for sharks via the possible reduction of gut and esophageal hooking. Additionally, on August 29, 2011, NMFS finalized a rule domestically implementing ICCAT Recommendation 10-07, which prohibits the retention, transshipping, landing, sorting, or selling of oceanic whitetip sharks caught in the HMS PLL fishery (76 FR

53652; August 29, 2011). Therefore, it can be reasonably assumed that prior to 2011 at least some individuals were retained in the HMS PLL fishery, whereas now all oceanic whitetips must be released with no more than 3 feet of trailing line. There is no basis to believe that the loss of 498 individuals over consecutive 3-year periods will reduce the distribution of the species. The species is widespread and wide-ranging, and the takes occur throughout the action area, not in any one area of their distribution.

Therefore, we conclude that the proposed action is not expected to have a population-level impact on the reproduction, numbers, or distribution of oceanic whitetip shark and 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. 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). Thus, 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 HMS PLL fishery (the subject of this 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. 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. Thus, management actions have helped to reduce the threat relative to past practices. The proposed action will also not impede the process of restoring the ecosystems that affect oceanic whitetip sharks.

While there is no recovery plan at this time, NMFS has developed a recovery outline to provide a preliminary strategy for recovery and conservation of the oceanic whitetip shark (https://www.fisheries.noaa.gov/resource/document/oceanic-whitetip-shark-recovery-outline). The recovery outline guides initial recovery actions while ensuring that future recovery options are not precluded due to a lack of interim planning. As such, this outline is meant to serve as an interim guidance document to direct recovery efforts, including recovery planning, for the oceanic whitetip shark until a full recovery plan is developed and approved. It presents a preliminary strategy for recovery of the species, as well as recommended high priority actions to stabilize and recover the species.

In advance of an approved recovery plan, the initial focus of the interim recovery program will be two-fold: 1) to stabilize population trends through reduction of threats, such that the species is no longer declining throughout a majority of its range and 2) to gather additional information through research and monitoring on the species' current distribution and abundance; reproductive periodicity and seasonality; location of breeding and nursery grounds; and mortality rates in commercial fisheries (including at-vessel and post-release mortality). Because the oceanic whitetip shark largely occurs in waters outside of U.S. jurisdiction, international coordination will be critical to ensuring recovery of the species. Therefore, to be effective, recovery actions would need to be undertaken throughout the species' range, both domestically and internationally. As detailed previously, the proposed action is not expected to have a population-level impact on oceanic whitetip sharks and thus would not impede the first goal of the recovery outline. The recovery outline lists maintaining existing U.S. laws and regulations that protect sharks and prohibit retention of oceanic whitetip sharks, including the regulations applicable to the HMS PLL fishery, as part of the recovery strategy, and the proposed action continues all current regulations. The second goal, to gather additional information, would not be impeded by the proposed action, and could benefit from observer information on interactions. For all of these reasons, 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. Therefore, the proposed action is not likely to jeopardize the continued existence of the species.

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 sperm whales, sea turtles, giant manta rays, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, or oceanic whitetip sharks. Since no critical habitat will be adversely affected, the action is not likely to destroy or adversely modify designated critical habitat.

Sperm Whales

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of sperm whales in the wild. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of the sperm whales.

Sea Turtles

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of ESA-listed sea turtle 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, hawksbill, or olive ridley sea turtles.

Giant Manta Ray

The proposed action is not expected to appreciably reduce the likelihood of survival and recovery of giant manta ray in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of the giant manta ray.

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 the Central and Southwest Atlantic DPS of scalloped hammerhead shark 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 oceanic whitetip shark in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of the oceanic whitetip shark.

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 reasonable and prudent measures (RPMs) and the terms and conditions of the incidental take statement (ITS) of the Opinion.

The take of the giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and the oceanic whitetip shark by the proposed action is not prohibited, as no Section 4(d) Rules for these species have been promulgated. However, a recent circuit court case held that non-prohibited incidental take must be included in the ITS.[16] 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.

9.1 Anticipated Incidental Take

NMFS anticipates the following incidental takes of sperm whale, NWA DPS loggerhead sea turtle, NA DPS green sea turtle, SA DPS green sea turtle, hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, olive ridley sea turtle, giant manta ray, Central and Southwest Atlantic DPS scalloped hammerhead shark, and oceanic whitetip shark may occur in the future because of the proposed action.

The level of takes occurring in the fishery is typically highly variable over time and across the action area. Factors such as water temperatures, currents and eddies, localized species abundances, and other factors that cannot be predicted, can all impact take levels. Because of this variability, take of any given species in any given year can vary widely, with some years well above, and other years well below, the average. 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 rolling time period is appropriate for meaningful monitoring of take and compliance with the ITS, except for sperm whales for which take is based on a 10-year rolling time period as detailed above, not static 10-year periods (i.e., 2020-2029, 2021-2030, 2022-2031, and so on, not 2020-2029, 2030-2039, and so on). 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., 2020-2022, 2021-2023, 2022-2024, 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 take estimates. Take up to the total take number and take up to the total mortality number in Table 9.1 is authorized under the ITS.

For sperm whales, the incidental take authorization and the RPMs and Terms and Conditions in the ITS are not in effect at this time because the incidental take of sperm whales has not been authorized under section 101(a)(5)(E) of the Marine Mammal Protection Act (MMPA) (see 16 U.S.C. 1536(b)(4)(C)). Following issuance of such authorization, the incidental take authorization and the RPMs and Terms and Conditions in the ITS take effect and become fully operative. Once in effect, the ITS covers running 10-year time periods, subject to the continued authorization of the incidental take under MMPA Section 101(a)(5)(E). NMFS may revise the ITS based on future MMPA incidental take authorizations.

[16] *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.

For the period before the ITS takes effect, NMFS is including a numerical reinitiation trigger. Reinitiation will occur if the annual average interactions with the northern Gulf of Mexico stock of sperm whale associated with this fishery is greater than or equal to 0.3 whales during future static, 5-year timeframes. Based on the jeopardy analysis above, NMFS has concluded that lethal take of the northern Gulf of Mexico sperm whales at a number that is less than or equal to this trigger would not reduce appreciably the likelihood of both survival and recovery of the sperm whale.

To authorize the incidental take under the MMPA section 101(a)(5)(E), among other findings, NMFS must determine that the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock (MMPA Section 101(a)(5)(E)(i)(I)). This is referred to as a negligible impact determination (NID). As is noted above, we anticipate that the fishery will interact with the northern Gulf of Mexico stock of sperm whales. NMFS has completed preliminary calculations of the impact of the fishery on this stock based on the best available scientific information. We believe the incidental take associated with this fishery as analyzed in this Opinion meets the criteria for issuance of a NID for the northern Gulf of Mexico sperm whale stock. Therefore, we are beginning the process of making a NID for the Gulf of Mexico sperm whale stock for the HMS PLL fishery, and including the HMS PLL fishery on the list of fisheries that will have a negligible impact on this stock and authorizing the incidental take pursuant to MMPA Section 101(a)(5)(E).

Note that for the sea turtle species, the mortalities are based on combined dead-on-retrieval numbers plus post-release mortality. Likewise for the Central and Southwest DPS of scalloped hammerhead shark and oceanic whitetip shark, the mortalities are based on combined dead-on-retrieval numbers plus post-release mortality. For the shark species, the lethal take that will be monitored and reported is dead-on-retrieval only as currently there is no way to monitor their post-release mortalities. When evaluating compliance with the ITS, post-release mortality of these shark species will be estimated by applying the anticipated post-release mortality rate discussed in Section 5 to the reported non-lethal interactions. We did not expect post-release mortalities of giant manta ray, as described in Section 5.

Table 9.1 Anticipated Future Take Estimates for the Proposed Action

	Total Takes	Mortalities
Sperm Whale (10-year)	3	3
Leatherback Sea Turtle (3-year)	996	275 (13+262)*
Loggerhead Sea Turtle (3-year)	1,080	280 (6+274)*
"Other Hardshell" Sea Turtle (any combination of NA green, SA green, hawksbill, Kemp's ridley, or olive ridley sea turtles) (3-year)	21	8 (3+5)*
Giant Manta Ray (3-year)	366	6
Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS (3-year)	576	249 (165+84)**
Oceanic Whitetip Shark (3-year)	1,362	498 (279+219)**

*Sea turtle mortalities include dead-on-retrieval and estimates of post-release mortality based on observed hooking location and gear removal. Numbers in parentheses are dead-on-retrieval + post-release mortality.

** Scalloped hammerhead and oceanic whitetip mortalities include dead-on-retrieval and estimates of post-release mortality based on post-release mortality rate from scientific literature, discussed in Section 5. Numbers in parentheses are dead-on-retrieval + post-release mortality.

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 DPSs: sperm whale, NWA DPS loggerhead sea turtle, NA DPS green sea turtle, SA DPS green sea turtle, hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, olive ridley sea turtle, giant manta ray, Central and Southwest Atlantic DPS scalloped hammerhead shark, and oceanic whitetip shark.

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. Sperm Whale, Sea Turtle, Giant Manta Ray, Scalloped Hammerhead Shark, and Oceanic Whitetip Shark Handling Requirements:
   Most, if not all, sperm whales, 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 sperm whales, sea turtles, giant manta rays, scalloped hammerhead sharks in the Central and Southwest DPS, and oceanic whitetip sharks following release. The SERO PRD Divisions shall advise the HMS Management Division how to ensure that fishermen in

the HMS PLL fishery receive relevant outreach materials and provide such materials describing how captured ESA-listed sea turtles and fish should be handled and how gear should be removed from ESA-listed sea turtles, fish, and marine mammals to minimize adverse effects from incidental take and reduce mortality. The HMS Management Division shall provide such training using materials provided by the SERO PRD Division to fishermen.

2. Monitoring the Frequency, Magnitude, and Details of Incidental Take:
The jeopardy analyses for sperm whales, sea turtles, giant manta rays, scalloped hammerhead sharks, and oceanic whitetip sharks 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. Additionally, to estimate sea turtle mortality, including post-release mortality, information is needed about hooking location and amount of gear remaining when the sea turtle is released. Therefore, the HMS Management Division must ensure that any takes of ESA-listed species are monitored and reported, coordinating with the SEFSC as necessary and appropriate. Such reports should allow the agency to: (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; (3) assess (for sea turtles) the hooking location and gear remaining on every sea turtle released to allow for post-release mortality estimations; and (4) detect when the level of anticipated take (lethal and non-lethal) 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.

Terms and Conditions Related to RPM #1:

1. The HMS Management Division must distribute outreach information to all HMS PLL 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. 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 and added to the HMS Management Division website.

2. The HMS Management Division must ensure that gear removal and sea turtle handling training consistent with the methodologies cited in #1 is available to new fishermen in the HMS PLL fishery, and that the certification program for participants in the fishery is maintained.

3. The HMS Management Division must coordinate with SERO PRD, within 30-days of issuance of this Opinion, to establish guidelines for gear removal in the case of sperm whale entanglement. When completed, this guidance must be distributed to all HMS PLL fisherman. In addition, the HMS Management Division must maintain a copy of this guidance on its website so that it is accessible to all HMS PLL fishermen. The HMS Management Division shall annually coordinate with SERO PRD to check for any updates to the guidance that may need to be distributed and added to its website.

4. The HMS Management Division must distribute the handling and release procedures for oceanic whitetip sharks and Central and Southwest Atlantic DPS of scalloped hammerhead sharks, available on the NMFS website, to all HMS PLL fishermen and must continue to maintain a copy of this guidance on its website so that it is accessible to all HMS PLL fishermen. The HMS Management Division shall annually coordinate with SERO PRD to check for any updates to the guidance that may need to be distributed and added to its website.

5. The HMS Management Division must distribute the handling and release procedures for giant manta rays to all HMS PLL fishermen, available at the NMFS website, and must continue to maintain a copy of this guidance on its website so that it is accessible to all HMS PLL 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 and added to its website.

6. The HMS Management Division must ensure that oceanic whitetip shark and the Central and Southwest Atlantic DPS of scalloped hammerhead shark and giant manta rays are included in applicable HMS educational and training workshops (i.e., Atlantic Shark Identification workshop, and Safe Handling, Release, and Identification workshop) required for dealers and permit holders participating in the HMS PLL fishery. The HMS Management Division must provide information on safe handling and release protocols, including any updates to these protocols, and information on status of these species under the ESA, in addition to any measures developed in the future related to reducing interactions with the species from PLL fishing gear. The Protected Resources Division will provide such information to the HMS Management Division and request it be provided at the workshops. In addition, for giant manta ray, the HMS Management Division must provide information on how to voluntarily report interactions (i.e., send an email to manta.ray@noaa.gov).

Terms and Conditions Related to RPM #2:

1. A minimum 8% observer coverage in each fishing year was established under the June 1, 2004 Opinion for the purpose of sea turtle observation. That 8% yearly average level is a minimum, not a target, and we note that observer coverage has typically been higher than 8% since implementation of the June 1, 2004 Opinion. However, with the inclusion of other ESA-listed species in this Opinion, the HMS Management Division must work with the appropriate observer program to ensure observer coverage in the HMS PLL fishery subject to this consultation is sufficient for monitoring take of those other ESA-listed species. Observer coverage must be maintained at a minimum of 8% as a yearly average as stipulated. NMFS (2004) recommends a level of observer coverage equal to that which provides estimates of a protected species interaction with an expected coefficient of variation (CV) of 30%. Since interactions with 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% for bycatch estimates for all of the species covered in the ITS 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.

2. The SEFSC must instruct POP observers to continue collecting detailed information on all sea turtle interactions, including initial interaction type, hooking location, amount of gear remaining upon release, and the animal's condition upon release, and the SEFSC must provide that information to the HMS Management Division. The HMS Management Division, in coordination with the SEFSC, must use this information to estimate the post-release mortality rate associated with the observed captures of the sea turtle species. Currently, the best scientific data available to estimate post-release mortality rate uses the species' release condition and applies that information to the post-release mortality criteria updated in Ryder et al. (2006) and the mortality tables revised by NMFS SEFSC (NMFS 2012d). If additional methods are developed and determined to be the best scientific data available, the HMS Management, SEFSC, and SERO PRD will coordinate to use those methods to estimate the post release mortality rate.

3. The HMS Management Division, in coordination with the SEFSC, must (1) collect and monitor observer and other reports from HMS targeted trips having sperm whale, sea turtle, giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark interactions, and (2) submit quarterly and annual reports detailing interactions with these species and the HMS pelagic longline fishery to SERO PRD, as described below.

4. On a quarterly basis, the HMS Management Division, in coordination with the SEFSC, must estimate and report to SERO PRD the total take and total mortalities (dead-on-retrieval and post-release mortality) of ESA-listed species in the HMS pelagic longline fishery. To estimate sea turtle post-release mortalities, use the process described in T&C 2, above, to estimate the post-release mortality rate and apply that rate to the total estimated non-lethal interactions. To estimate oceanic whitetip shark and the Central and Southwest Atlantic DPS of scalloped hammerhead shark post-release mortalities, use the estimated post-release mortality rate (20%) described in Section 5 above, based on Musyl and Gilman (2019), and apply that rate to the total estimated non-lethal interactions. If current effort data are not available, estimated total take and total mortalities can be prepared using prior effort estimates. These quarterly reports must be submitted no later than 45 days into the subsequent quarter. Reports must be sent to the NMFS Assistant Regional Administrator for Protected Resources, Southeast Regional Office, Protected Resources Division, 263 13th Avenue South, St. Petersburg, Florida 33701-5505; transmittal by email is acceptable.

3. On an annual basis, the HMS Management Division, in coordination with the SEFSC, must submit a report detailing interactions between ESA-listed species and the HMS pelagic longline fishery 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) *Sperm Whale Reports*: must include any information available on size (adult, juvenile, calf), time and location (i.e., lat./long. and fishery reporting area) of interaction, nature of the interaction, and status (i.e., dead, alive, injured) of the individual upon removal of gear.
  - (ii) *Sea Turtle Reports*: must include species, carapace length (curved or straight length must be noted), time and location (i.e., lat./long. and fishery reporting area) of capture, and status (i.e., alive, dead, injured) upon return to the water, in addition to the information specified in #2 above. Any additional information such as tags, etc. should also be reported.
  - (iii) *Giant Manta Ray 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 fishery reporting area) of capture, and status (i.e., dead, alive, injured) upon return to the water should be reported.
  - (iv) *Shark Reports*: for the Central and Southwest Atlantic DPS of 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.

(b) Information Required on Fishery Operations
Information on mainline length, sets, hooks fished per set, hook type, soak time, bait used, fishery reporting area must be included in the reports.

(c) Reports must also estimate the total take and total mortality (dead-on-retrieval and post-release mortality) of ESA-listed species in the HMS PLL fishery subject to this consultation, based on availability of effort data and reported and observed takes. To estimate sea turtle post-release mortalities, use the process described in T&C 2, above, to estimate the post-release mortality rate and apply that rate to the total estimated non-lethal interactions. To estimate oceanic whitetip shark and the Central and Southwest Atlantic DPS of scalloped hammerhead shark post-release mortalities, use the estimated post-release mortality rate (20%) described in Section 5 above, based on Musyl and Gilman (2019), and apply that rate to the total estimated non-lethal interactions. Total take and total mortality must be estimated over rolling 3-year periods, as described in the ITS. If the estimated take and/or mortality of sperm whales, sea turtles, giant manta rays, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, and oceanic whitetip sharks, 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. Annual reports for the previous year must be completed by June 30 each year.

(d) Annual reports must be sent to the NMFS Assistant Regional Administrator for Protected Resources, Southeast Regional Office, Protected Resources Division, 263 13th Avenue South, St. Petersburg, Florida 33701-5505; transmittal by email is acceptable.

10. 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.

1. The SEFSC should devise a probability-based approach or other statistical method to evaluate take in the HMS PLL fishery. Use of such a method, instead of using a single number to indicate exceedance of the ITS, may provide a better approach to evaluating the actual risk of greater than expected take levels occurring. Such an approach would allow NMFS to establish a trigger that reduces the likelihood of requiring reinitiation

unnecessarily because of inherent variability in take levels (which is expected to be large), but still allows for an accurate assessment of how the fishery is performing versus expectations. Once such a method is devised, SEFSC and SERO PRD would then consult to determine whether the new approach is biologically valid and equivalent to the current method, and provides a better tool for evaluating and managing takes in the HMS pelagic longline fishery.

2. In order to understand why post-release mortality rates of leatherback and loggerhead sea turtles were higher than expected following the 2004 Opinion, NMFS should evaluate available data on interactions (e.g., hooking location, gear removal). NMFS notes that even with the higher than expected post-release mortality rates, the lethal and non-lethal take anticipated in this Opinion is not likely to jeopardize the continued existence of these species.

3. In order to better understand sea turtle populations and the impacts of incidental take in the HMS PLL fishery, NMFS should support in-water abundance estimates of sea turtles to achieve more accurate status assessments for these species and improve our ability to monitor them.

4. 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 PLL fishery, as well as other, fisheries. This will help improve the accuracy of future assessments of the effects of different levels of take on sea turtle populations.

5. Given the ESA listings for the 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 as 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.

6. NMFS should expand and continue supporting research to better estimate giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark mortality, including dead-on-retrieval and post-release mortality, in the HMS PLL fishery.

7. NMFS should investigate best methods for handling, gear removal, and safe release of giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark in the HMS PLL fishery.

8. NMFS should conduct research on gear modifications to increase survivorship of giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark when caught in the HMS PLL fishery.

9. NMFS should survey HMS PLL fishermen regarding their experience and recommendations regarding the effectiveness of safe release techniques.

10. NMFS should conduct and/or fund research that will improve understanding of giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark population distribution, abundance, trends, and structure through research, monitoring, and modeling.

11. NMFS should conduct and/or fund research that will improve understanding of giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark reproductive periodicity and seasonality to inform future management measures for minimizing impacts to the species during key life history functions.

11. 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.

In addition to the reinitiation criteria provided above, we developed the following numerical reinitiation trigger for sperm whales, which applies in the time period before sperm whale take is authorized under the MMPA and the ITS specified above, including the RPMs and terms and conditions, take effect. The trigger is based on the annual average interactions for the past five years (2014-2018), and will be used as a trigger for reinitiation should this average be exceeded. The additional sperm whale trigger is as follows:

Sperm whale: 0.3 annual average interactions over future, static 5 year periods as long as the ITS above with respect to sperm whale is not in effect. If the ITS as to sperm whale is in effect, the numeric trigger is 3 takes, which can be lethal or non-lethal, over rolling 10-year periods.

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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-Non-Pelagic Longline [SER]	3 Years	91-No more than 51 lethal	7-No more than 4 lethal	22-No more than 11 lethal	NA DPS 46-No more than 25 lethal SA DPS 3-No more than 2 lethal	2-No more than 1 lethal
HMS-Pelagic Longline [SER] (prior to this consultation)	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	105-No more than 18 lethal for these species in combination	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

Appendix B. Incidental Takes Anticipated from SARBO

Table B1. Anticipated Future Take Per 3 Consecutive Year Period

Species	Nonlethal Take-Observed	Lethal Take-Observed	Lethal Take-Unobserved	Total Lethal Observed + Unobserved Take	Sea Turtle Lost Egg Clutch
Green Sea Turtle NA DPS	742	59	59	118	3
Green Sea Turtle SA DPS	40	4	4	8	0
Kemp's Ridley Sea Turtle	1,340	58	58	116	1
Leatherback Sea Turtle	369	0	4	4	6
Loggerhead Sea Turtle NWA DPS	5,270	107	107	214	65
Atlantic Sturgeon South Atlantic DPS	499	73	0	73	N/A
Atlantic Sturgeon Carolina DPS	319	47	0	47	N/A
Atlantic sturgeon Chesapeake Bay DPS	91	14	0	14	N/A
Atlantic Sturgeon New York Bight DPS	34	5	0	5	N/A
Atlantic Sturgeon Gulf of Maine DPS	1	1	0	1	N/A
Shortnose sturgeon	6	8	6	14	N/A
Giant manta ray	89	0	0	0	N/A

Table B2. Anticipated Future Take Per Other Defined Time Period

Species	Nonlethal Take-Observed	Lethal Take-Observed	Lethal Take- Unobserved
Smalltooth sawfish (U.S. DPS)	1 total per 3 year period	1 total per 9 year period[17]	
Elkhorn Coral	2 total per 10 year period	1 total per 10 year period	Monitoring required = no unobserved
Staghorn coral	1,105 total per 10 year period	195 total per 10 year period	Monitoring required = no unobserved
Lobed star coral	43 total per 10 year period	8 total per 10 year period	Monitoring required = no unobserved
Mountainous star coral	136 total per 10 year period	25 total per 10 year period	Monitoring required = no unobserved
Boulder star coral	63 total per 10 year period	11 total per 10 year period	Monitoring required = no unobserved

[17] For smalltooth sawfish, a total of 3 takes in authorized every 9 years, with up to 1 lethal take.