Endangered Species Act (ESA) - Section 7 Consultation
Biological Opinion

Action Agency: National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office (SERO), St. Petersburg, Florida

Activity: Reinitiation of Endangered Species Act (ESA) Section 7 Consultation on the Implementation of the Sea Turtle Conservation Regulations under the ESA and the Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act (MSFMCA)

Consulting Agency: NOAA, NMFS, SERO, Protected Resources Division (F/SER3) and Sustainable Fisheries Division (F/SER2)

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Table of Contents

Figures

Tables

Acronyms and Abbreviations

BIRNM	Buck Island Reef National Monument
BOEM	Bureau of Ocean Energy Management
CCL	curved carapace length
CPUE	catch per unit effort
DDT	dichlorodiphenyltrichloroethane
DNA	deoxyribonucleic acid
DO	dissolved oxygen
DPS	distinct population segment
DTRU	Dry Tortugas Recovery Unit

DWH	DEEPWATER HORIZON
EEZ	exclusive economic zone
ESA	Endangered Species Act
FMP	fishery management plan
FP	Fibropapillomatosis
F/SER2	(NMFS SERO) Sustainable Fisheries Division
F/SER3	(NMFS SERO) Protected Resources Division
FWC	Florida Fish and Wildlife Conservation Commission
GADNR	Georgia Department of Natural Resources
GARFO	(NMFS) Greater Atlantic Regional Fisheries Office
GCRU	Greater Caribbean Recovery Unit
GMFMC	Gulf of Mexico Fishery Management Council
GRBO	Gulf of Mexico Regional Biological Opinion
HAB	harmful algal bloom
HMS	highly migratory species
IPCC	Intergovernmental Panel on Climate Change
ITS	incidental take statement
LDWF	Louisiana Department of Wildlife and Fisheries
MSA	mixed stock analysis
MSFCMA	Magnuson Stevens Fishery Conservation and Management Act
NA	North Atlantic (Ocean)
NCWRC	North Carolina Wildlife Resources Commission
NEAMAP	Northeast Area Monitoring and Assessment Program
NEFSC	(NMFS) Northeast Fisheries Science Center
NGMRU	Northern Gulf of Mexico Recovery Unit
NLAA	may affect, not likely to adversely affect
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPS	U.S. National Park Service
NRU	Northern Recovery Unit
NWA	Northwest Atlantic Ocean
PAH	polycyclic aromatic hydrocarbons
PAIS	Padre Island National Seashore
PCB	polychlorinated biphenyls
PFRU	Peninsular Florida Recovery Unit
PIM	post-interaction mortality
PVA	population viability analysis
RPMs	reasonable and prudent measures
SA	South Atlantic (Ocean)
SCL	straight carapace length
SD	standard deviation
SAFMC	South Atlantic Fishery Management Council
SCDNR	South Carolina Department of Natural Resources

SEFSC	(NMFS) Southeast Fisheries Science Center
SERO	(NMFS) Southeast Regional Office
STSSN	Sea Turtle Stranding and Salvage Network
TED	turtle excluder device
TEWG	Turtle Expert Working Group
TL	total length
USACE	U.S. Army Corps of Engineers
USCG	U.S. Coast Guard
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
YOY	young-of-year

Units of Measurement

°C	degree Celsius
°F	degree Fahrenheit
°N	degree north (latitude)
cm	centimeter
ft	feet
in	inch
kg	kilogram
lb	pound
m	meter
mm	millimeter
nm	nautical mile
oz	ounce

INTRODUCTION

Section 7(a)(2) of the ESA of 1973, as amended (16 U.S.C. § 1531 *et seq*.), requires each federal agency to "insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species." Section 7(a)(2) requires federal agencies to consult with the appropriate Secretary on any such action. We, along with the U.S. Fish and Wildlife Service (USFWS), share responsibilities for administering the ESA.

Consultation is required when a federal action agency determines that a proposed action "may affect" listed species or designated critical habitat. Consultation is concluded after we determine the 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 critical habitat. The Opinion states the amount or extent of incidental take of the listed species that may occur, develops measures (i.e., reasonable and prudent measures [RPMs]) to reduce the effect of take, and recommends conservation measures to further the recovery of the species. Notably, no incidental destruction or adverse modification of designated critical habitat can be authorized, and thus there are no RPMs—only reasonable and prudent alternatives that must avoid destruction or adverse modification.

This document represents our Opinion on the effects of the implementation of the sea turtle conservation regulations applicable to shrimp trawling and the authorization of southeast U.S. shrimp fisheries in federal waters on threatened and endangered species and designated critical habitat, in accordance with Section 7 of the ESA. This Opinion is the result of an intra-agency consultation with our Sustainable Fisheries Division (F/SER2). For the actions described in this document, we are both the action agency (F/SER2) under our authorities to conserve sea turtles under the ESA and to manage federal shrimp fishing under the Magnuson Stevens Fishery Conservation and Management Act (MSFCMA) (16 U.S.C. §1801 *et seq*.), and the consulting agency (F/SER3). There is no applicant associated with this proposed action.

1 CONSULTATION HISTORY

We have conducted Section 7 consultation on our sea turtle conservation regulations governing the use of Turtle Excluder Devices (TEDs) and the authorization of southeast U.S. shrimp fisheries in federal waters numerous times over the years (e.g., NMFS 1987; NMFS 1992; NMFS 1994; NMFS 1996; NMFS 1998; NMFS 2002a; NMFS 2005a; NMFS 2006; NMFS 2012a). The details and history of consultation documented in these past Opinions are incorporated herein by reference. The last time Section 7 consultation was conducted on the implementation of the sea turtle conservation regulations under the ESA and the authorization of the southeast U.S. shrimp fisheries in federal waters under the MSFCMA was in April 2014 (NMFS 2014). The 2014 Opinion concluded the fisheries were not likely to jeopardize the continued existence of green (both the Florida breeding population and non-Florida breeding population), hawksbill, leatherback, Kemp's ridley, or loggerhead sea turtles (the Northwest

Atlantic Ocean [NWA] distinct population segment [DPS]), as well as Atlantic sturgeon (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, or South Atlantic [SA] DPSs), Gulf sturgeon, or smalltooth sawfish (U.S. DPS).

The 2014 Opinion noted it was not possible to reliably quantify the anticipated amount of take of sea turtles in the southeast U.S. shrimp fisheries, and provided an explanation why it was not possible using the best available information at the time. It further explained the numerical take estimates generated in that Opinion were "unacceptably uncertain to rely on them extensively in analyzing impacts…." Therefore, the 2014 Opinion relied on fisheries effort and compliance with the TED requirements to monitor compliance with the incidental take statement (ITS).

As provided in 50 CFR 402.16, reinitiation of formal consultation is required if discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: 1) The amount or extent of the taking specified in the ITS is exceeded; 2) new information reveals effects of the action that may affect listed species or critical habitat (when designated) in a manner or to an extent not previously considered; 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 identified action.

On July 1, 2016, we received a request from F/SER2 for Section 7 reinitiation on the implementation of the sea turtle conservation regulations under the ESA and the authorization of the southeast U.S. shrimp fisheries in federal waters under the MSFCMA due to the ESA listings of the North and South Atlantic green sea turtle DPSs. The request also included a determination that the ongoing action would not violate Sections 7(a)(2) and 7(d) of the ESA during the reinitiation period.

On December 14, 2016, we received a request from F/SER2 for Section 7 reinitiation on the implementation of the sea turtle conservation regulations under the ESA and the authorization of the southeast U.S. shrimp fisheries in federal waters under the MSFCMA due to the ESA listing of Nassau grouper. The request also included a determination that the ongoing action would not violate Sections 7(a)(2) and 7(d) of the ESA during the reinitiation period.

On June 17, 2019, we received a request from F/SER2 for Section 7 reinitiation on the implementation of the sea turtle conservation regulations under the ESA and the authorization of the southeast U.S. shrimp fisheries in federal waters under the MSFCMA due to the ESA listings of giant manta ray and Gulf of Mexico Bryde's whale.[1] The request also included a determination that the ongoing action would not violate Sections 7(a)(2) and 7(d) of the ESA during the reinitiation period.

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[1] Recently, the Gulf of Mexico Bryde's whale was identified as a new species known as Rice's whale (*Balaenoptera ricei*), but the taxonomy has not been officially recognized yet.

Since the 2014 Opinion, we have developed new bycatch information to better analyze the effects of the shrimp fisheries on sea turtle populations, we have issued a final rule requiring TEDs for a portion of the skimmer trawl fisheries, and we have listed new species under the ESA (i.e., Bryde's whale, giant manta ray, green sea turtle DPSs, and Nassau grouper). For these reasons, we are reinitiating Section 7 consultation. Consequently, this Opinion supersedes the 2014 Opinion and fulfills our Section 7 consultation responsibilities on both our implementation of the existing sea turtle conservation regulations under the ESA, and our authorization of federal shrimp trawling under the MSFCMA for all listed species.

2 DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA

2.1 Proposed Action

We are proposing to continue to: 1) conserve sea turtles via our sea turtle conservation regulations under the ESA for the southeast U.S. shrimp fisheries, which involve extending regulatory authorization to incidentally take sea turtles, subject to specific conditions; and 2) authorize shrimp trawling in the Exclusive Economic Zone (EEZ) under the Gulf of Mexico and South Atlantic Fishery Management Councils' Shrimp Fishery Management Plans (FMPs).

Sea Turtle Conservation Regulations

Current sea turtle conservation regulations for the southeast U.S. shrimp fisheries are detailed at 50 CFR 222, 50 CFR 223.205-207, and 50 CFR 224.104(a), all of which are incorporated herein by reference. The implemented focus of our conservation regulations for the shrimp fisheries, however, relate to TED requirements for otter trawl vessels and associated exemptions for all other shrimp trawlers. Our definition of "shrimp trawler" is any vessel that is equipped with one or more trawl nets and that is capable of, or used for, fishing for shrimp, or whose on-board or landed catch of shrimp is more than 1%, by weight, of all fish comprising its on-board or landed catch. The incidental taking of sea turtles during shrimp trawling is exempted from the taking prohibition of Section 9 of the ESA if the conservation measures specified in the sea turtle conservation regulations (50 CFR 223) are followed. The regulations require most shrimp trawlers operating in the southeast U.S. (Atlantic and Gulf areas; see 50 CFR 223.206) to have an approved TED installed in any net that is rigged for fishing to provide for the escape of sea turtles. TEDs incorporate an escape opening, usually covered by a webbing flap, which allow sea turtles to escape from trawl nets. To be an approved TED, the design must be shown to be 97% effective in excluding sea turtles during testing based upon specific testing protocols (50 CFR 223.207(e)(1)). We have approved several TED designs, including single-grid hard TEDs, hooped hard TEDs conforming to a generic description, and one type of soft TED—the Parker soft TED (see 50 CFR 223.207).

We have established exemptions to the TED requirements on the basis that the exempted activities did not present a threat to sea turtle populations. Generally, vessels that: have no power or mechanical-advantage trawl retrieval system; are bait shrimpers that retain all live shrimp on board with a recirculating seawater system; fish with a pusher-head trawl, skimmer trawl, or wing net; or use a single try net with a headrope 12 feet (ft) or less in length, may currently use alternative tow times in lieu of TEDs. Additionally, we exempted beam or roller trawls and shrimp trawlers fishing for royal red shrimp (a deep-water shrimp species) from the TED requirements. The alternative tow time restrictions specify tow times are not to exceed 55 minutes from April 1 through October 31, and 75 minutes from November 1 through March 31 (50 CFR 223.206(d)(3)(i)(A) and (B)).

On February 21, 2003, we issued a final rule (68 FR 8456) amending the sea turtle conservation regulations to protect large loggerhead, green, and leatherback sea turtles. This final rule required all nets rigged for fishing on shrimp trawlers operating in the offshore waters of the southeast U.S., as well as the inshore waters of Georgia and South Carolina, to use either a double cover flap TED, a single-grid hard TED with a 71-inch (in) opening, or a Parker soft TED with a 96-in opening. As alluded to in Section 2, we also published a final rule on December 20, 2019 (84 FR 70048), that would withdraw the alternative tow times and require TEDs on skimmer trawl vessels 40 ft and greater in length effective April 1, 2021; skimmer trawl vessels less than 40 ft in length would be required to continue to comply with alternative tow time requirements per 50 CFR 223.206(d)(2)(ii)(A)(*3*). On March 31, 2021 (86 FR 16676), we delayed the effective date of this final rule until August 1, 2021, due to safety and travel restrictions related to the COVID-19 pandemic that prevented necessary training and outreach for fishers.

Our sea turtle conservation regulations under the ESA apply to all shrimp trawlers, wherever they occur. As such, they apply in federal waters (i.e., the Gulf and South Atlantic EEZ) where we authorize shrimp trawling via two FMPs under the MSFCMA, and in state waters, where respective state resource agencies authorize their fisheries. Section 4 (d) of the ESA allows us to issue regulations for threatened species as deemed necessary and advisable for the conservation of such species. Section 11(f) of the ESA allows us to promulgate such regulations that may be appropriate to enforce the ESA. For example, although we do not authorize state fisheries, we do mandate that affected state-authorized fisheries comply with our promulgated sea turtle conservation regulations, which require most shrimp trawlers to use TEDs or comply with alternative tow-time restrictions. In turn, these regulations provide an exemption from the Section 9 take prohibitions that would otherwise apply to these fisheries.

Shrimp Trawling

As mentioned above, we authorized shrimp trawling in the EEZ via the Gulf of Mexico Fishery Management Council's (GMFMC) FMP for the Shrimp Fishery of the Gulf of Mexico, U.S. Waters (GMFMC 1981) and the South Atlantic Fishery Management Council's (SAFMC) FMP for the Shrimp Fishery of the South Atlantic Region (SAFMC 1993). A complete description of the federal shrimp fisheries can be found in these FMPs and their subsequent plan amendments, as well as in a final environmental impact statement (FEIS) we published analyzing the effects of the skimmer trawl final rule (NMFS 2019a), all of which are incorporated herein by reference. A summary of this information follows.

The Northern Gulf of Mexico (and North Carolina) shrimp fisheries are based primarily on 2 species, brown shrimp (*Farfantepenaeus aztecus*) and white shrimp (*Litopenaeus setiferus*). The southeast U.S. shrimp fisheries also include pink shrimp (*Farfantepenaeus duorarum*) and royal red shrimp (*Hymenopenaeus robustus*), while seabobs (*Xiphopenaeus kroyeri*) and rock shrimp (*Sicyonia brevirostris*) generally occur as incidental catch.

More than half of the commercial shrimp vessels fall into a size range from 56 to 75 ft (GMFMC 2016). Federal permits for shrimp vessels are currently required, and state license requirements vary. A moratorium on federal shrimp permits was approved by the GMFMC in 2005. Many vessels maintain licenses in several states because of their migratory fishing strategy. The number of vessels in the shrimp fisheries at any one time varies due to economic factors such as the price and availability of shrimp and cost of fuel.

As of March 16, 2021, there were 1,216 valid or renewable moratorium permits for the federal Gulf of Mexico shrimp fishery (SPGM), which is a significant decline from the 2,385 permits encompassed by a previously open-access Gulf of Mexico federal shrimp fishery, which sunset on March 25, 2007 (NMFS statistics). Additionally, there are 301 current Gulf of Mexico royal red shrimp endorsements, which must be accompanied by a valid SPGM permit. In the South Atlantic, there were 439 federally-permitted (open-access) vessels in the penaeid shrimp fishery, 119 (open-access) permits for the Carolina Zone rock shrimp fishery, and 95 valid (limited-access) permits for the South Atlantic EEZ rock shrimp fishery.

Various types of gear are used to capture shrimp, including but not limited to: cast nets, dip nets, haul seines, otter trawls, stationary butterfly nets, wing nets (butterfly trawls), skimmer trawls, traps, and beam trawls. The otter trawl, with various modifications, is the dominant gear used in offshore waters. A basic otter trawl consists of a heavy mesh bag with wings on each side designed to funnel the shrimp into the "cod end" or "tail bag." A pair of otter boards or trawl doors positioned at the end of each wing hold the mouth of the net open by exerting a downward and outward force at towing speed. A lead line or footrope extends from door to door on the bottom of the trawl, while a cork line or headrope is similarly attached at the top of the net. A "tickler chain" is also attached between the trawl doors that runs just ahead of the net, and is used to spook shrimp off the bottom and into the trawl net. The lead lines of larger nets are weighted with a 1/4-to 3/8-in loop chain attached at about 1-ft intervals with a 14- to 16-in drop. Many larger nets are also equipped with rollers on the lead line that keeps the lead line from digging into muddy bottom.

Shrimp trawl nets are usually constructed of nylon or polyethylene mesh webbing, with individual mesh sizes ranging from as small as 1-1/4 in to 2 in. The sections of webbing are assembled according to the size and design (usually flat, balloon, or semi-balloon) of trawl desired, which affects the width and height of the trawl's opening and its bottom-tending characteristics. The tongue or "mongoose" design incorporates a triangular tongue of additional webbing attached to the middle of the headrope pulled by a center towing cable, in addition to

the 2 cables pulling the doors. This configuration allows the net to spread wider and higher than conventional nets and as a result has gained much popularity for white shrimp fishing.
Until the late 1950s, most shrimp vessels pulled single otter trawls, ranging from 80 to 100 ft in width, directly astern of the boat. Double-rig trawling was introduced into the shrimp fleet during the late 1950s. The single large trawl was replaced by 2 smaller trawls, each 40 to 50 ft in width, towed simultaneously from stoutly constructed outriggers located on the port and starboard sides of the vessels. The advantages of double-rig trawling include: (1) increased catch per unit of effort, (2) fewer handling problems with the smaller nets, (3) lower initial gear costs, (4) a reduction in costs associated with damage or loss of the nets, and (5) greater crew safety.

In 1972, the quad rig was introduced in the shrimp fisheries, and by 1976 it became widely used in the EEZ of the western Gulf of Mexico. The quad rig consists of a twin trawl pulled from each outrigger (i.e., 4 trawl nets). One twin trawl typically consists of two 40- or 50-ft trawls connected to a center sled and spread by 2 outside trawl doors. Thus, the quad rig with 2 twin trawls has a total spread of 160-200 ft versus the total spread of 110 ft in the old double rig of two 55-ft trawls. The quad rig has less drag and is more fuel efficient. The quad rig is the primary gear used in federal waters by larger vessels. Smaller boats and inshore trawlers often still use single- or double-rigged nets.

Try nets are small otter trawls about 12 to 16 ft in width that are used to test areas for shrimp concentrations. These nets are towed during regular trawling operations and lifted periodically to allow the fishers to assess the amount of shrimp and other fish and shellfish being caught. These amounts in turn determine the length of time the large trawls will remain set or whether more favorable locations will be selected. Try nets with a headrope length greater than 12 ft are required to use TEDs, while try nets 12 ft or less are required to comply with alternative tow times if no TED is installed (per 50 CFR 223.206(d)(2)(ii)(A)(*5*)).

Wing nets (butterfly trawls or "paupiers") were introduced in the 1950s and used on shrimp boats either under power or while anchored. A butterfly trawl consists of square metal frame which forms the mouth of the net. Webbing is attached to the frame and tapers back to a cod end on either side of the vessel. The vessel is then anchored in tidal current or the nets are "pushed" through the water by the vessel. Louisiana also licenses the use of stationary wing nets, which typically consist of a single net attached to a platform and is tended while it fishes, similar to a channel net used in North and South Carolina; the majority of licensed wing nets in Louisiana are associated with stationary platforms or docks. There is also a unique wing net fishery that primarily operates in Biscayne Bay, Miami-Dade County, Florida, sight-targeting pink shrimp at night. These vessels use light monofilament webbing that fish the surface when shrimp are abundant, typically around the full moon (Johnson et al. 2012).

Vietnamese fishers began moving into Louisiana in the early 1980s and introduced a gear called the "xipe" or "chopstick" net around 1983. The chopstick was attached to a rigid or flexible frame similar to the wing net; however, the frame mounted on the bow of the boat was attached to a pair of skids and fished by pushing the net along the bottom. As with wing nets, the contents of the net could be picked up and dumped without raising the entire net out of the water as is necessary with an otter trawl.

The skimmer trawl was developed for use in some areas primarily to catch white shrimp, which have the ability to jump over the cork line of standard trawls while being towed in shallow water. The skimmer net frame allows the net to be elevated above the water while the net is fishing, thus preventing shrimp from escaping over the top. Owing to increased shrimp catch rates, less debris or bycatch, and lower fuel consumption than otter trawlers, the use of skimmer trawls quickly spread in several coastal states. Within the Gulf of Mexico, Louisiana, Mississippi, Alabama, and Florida include skimmer trawls as an allowable gear. In the South Atlantic, North Carolina is the only state that permits skimmer trawl gear.

The basic components of a skimmer trawl include a frame, the net, heavy weights, skids or "shoes," and tickler chains. The net frame is usually constructed of schedule 80 steel or aluminum pipe or tubing and is either L-shaped (with an additional stiff leg) or a trapezoid design. When net frames are deployed, they are aligned perpendicularly to the vessel and cocked or tilted forward and slightly upward. This position allows the net to fish better and reduces the chance of the leading edge of the skid digging into the bottom and subsequently damaging the gear. The frames are maintained in this position by 2 or more stays or cables to the bow. The outer leg of the frame is held in position with a "stiff leg" to the horizontal pipe and determines the maximum depth at which each net is capable of working. To the bottom of the outer leg is attached the skid or "shoe," which allows the frame to ride along the bottom, rising and falling with the bottom contour. Tickler chains and lead lines comprise the bottom of this gear.

In 2007, the Southeast Fisheries Science Center (SEFSC) initiated a mandatory observer program for the commercial shrimp fishery operating in the United States Gulf of Mexico through Amendment 13 to the Gulf of Mexico Shrimp FMP. In 2008, the SEFSC expanded the observer program to include the penaeid and rock shrimp fisheries in the South Atlantic through Amendment 6 to the Shrimp FMP for the South Atlantic Region. These observer requirements were established under the authority provided by the MSFCMA; thus, the requirements are limited to vessels with federal fishing permits issued under the respective FMPs. Logbooks are not required on every vessel, but the SEFSC selects a random sample of vessels each year to carry observers and to use electronic logbooks. Additionally, in the Gulf of Mexico a vessel and gear characterization form must be completed and submitted annually, and a voluntary component of the observer program continues for the purposes of bycatch reduction device development and evaluation.

In addition to the MSFCMA based programs, observers may also be required on shrimp vessels pursuant to the authority in the ESA and the Marine Mammal Protection Act. Unlike the MSFCMA authority, these programs are not limited to federally permitted vessels. The ESA authority has been used most recently to require observers on skimmer trawls that fish almost exclusively in state waters.

While our sea turtle conservation regulations are permanent (i.e., barring regulatory repeal or amendment) and the federal shrimp fisheries are expected to continue into the foreseeable future, for purposes of this Opinion we will only consider the effects of the proposed action within the next 10 years. We have opted to limit the lifespan of this Opinion to 10 years due to the complexity of forecasting the potential effects of climate change over a longer timeline (these issues are discussed in more detail in Sections 3.2.1 and 4.4). Furthermore, potential changes to sea turtle populations (for example) and the effects of the fisheries on those increasing or decreasing population sizes could cast doubts on our conclusions over a longer time-period; we do not believe we can reliably evaluate the impact of these effects over a longer time frame. We believe, however, that 10 years presents a reasonable time-period to forecast both the effects of climate change and the effects of the action on affected ESA-listed species that would avoid unnecessary speculation and ensure our jeopardy conclusions in this Opinion remain valid. Despite the 10 year limitation on the lifespan of this Opinion, our analysis of effects does consider the effects of these actions occurring within this 10-year time frame with longer lasting impacts that may extend beyond the 10-year period. For example, the death of long-lived mature sea turtles may have population level effects that extend well beyond the 10 years, and those effects are appropriately incorporated into the analysis.

2.2 Action Area

The action area for this consultation includes the Gulf and South Atlantic EEZ, and adjacent marine and tidal state waters of the Gulf and South Atlantic area (i.e., from the Mexico-Texas border to the North Carolina-Virginia border). The Gulf EEZ extends from 9 nautical miles (nm) seaward of the states of Florida and Texas, and 3 nm seaward of the states of Alabama, Mississippi, and Louisiana, out to 200 nm from the baseline from which the territorial sea of the United States is measured. The South Atlantic EEZ extends from 3 nm seaward of the states of North Carolina, South Carolina, Georgia, and Florida, out to 200 nm from the baseline from which the territorial sea of the United States is measured.

3 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

Table 1 below documents all listed species that may occur within the action area, while Table 2 documents all critical habitat within the action area.

Table 1. Status of Listed Species in the Action Area (E= Endangered, T=Threatened). Green cells represent species we believe will not be adversely affected by the proposed action, which are discussed in more detail in the accompanying text in this section.

Species		Scientific Name	Status	Geographic Area
Whales	Sei whale	*Balaenoptera borealis*	E	South Atlantic
	Blue whale	*Balaenoptera musculus*	E	South Atlantic, EEZ only
	Fin whale	*Balaenoptera physalus*	E	South Atlantic
	North Atlantic right whale	*Eubalaena glacialis*	E	South Atlantic
	Sperm whale	*Physeter macrocephalus*	E	South Atlantic and Gulf, EEZ only
	Humpback whale	*Megaptera novaeangliae*	E	South Atlantic

	Bryde's whale	*Balaenoptera edeni* ssp.	E	Gulf
Sea Turtles	Loggerhead sea turtle, NWA DPS	*Caretta caretta*	T	South Atlantic and Gulf
	Green sea turtle, NA and SA DPSs	*Chelonia mydas*	T	South Atlantic and Gulf
	Leatherback sea turtle	*Dermochelys coriacea*	E	South Atlantic and Gulf
	Hawksbill sea turtle	*Eretmochelys imbricata*	E	South Atlantic and Gulf
	Kemp's ridley sea turtle	*Lepidochelys kempii*	E	South Atlantic and Gulf
Fish	Shortnose sturgeon	*Acipenser brevirostrum*	E	South Atlantic, within state waters only
	Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	E/T[2]	South Atlantic
	Gulf sturgeon	*Acipenser oxyrinchus desotoi*	T	Gulf
	Smalltooth sawfish, U.S. DPS	*Pristis pectinata*	E	South Atlantic and Gulf
	Nassau grouper	*Epinephelus striatus*	T	South Atlantic and Gulf
	Oceanic whitetip shark	*Carcharhinus longimanus*	T	South Atlantic and Gulf
	Giant manta ray	*Manta birostris*	T	South Atlantic and Gulf
Corals	Staghorn coral	*Acropora cervicornis*	T	South Atlantic and Gulf
	Elkhorn coral	*Acropora palmata*	T	South Atlantic and Gulf
	Pillar coral	*Dendrogyra cylindrus*	E	South Atlantic
	Lobed star coral	*Montastraea annularis*	E	South Atlantic and Gulf
	Mountainous star	Montastraea faveolata	E	South Atlantic and Gulf
	Knobby star coral	*Montastraea franksi*	E	South Atlantic and Gulf
	Rough cactus coral	*Mycetophyllia ferox*	E	South Atlantic and Gulf
	Lamarck's sheet coral	*Agaricia lamarcki*	T	South Atlantic and Gulf
	Elliptical star coral	*Dichocoenia stokesii*	T	South Atlantic and Gulf
Plants	Johnson's seagrass	*Halophila johnsonii*	T	South Atlantic, within state waters only

Table 2. Designated Critical Habitat in the Action Area. Green cells represent critical habitat we believe will not be adversely affected by the proposed action, which are discussed in more detail in the accompanying text in this section.

	Species	Geographic Area
Critical Habitat	Loggerhead sea turtle, NWA DPS	South Atlantic and Gulf
	North Atlantic right whale	South Atlantic
	Gulf sturgeon	Gulf, within state waters only
	Smalltooth sawfish	South Atlantic, within shallow state waters only
	Elkhorn and staghorn corals	South Atlantic
	Johnson's seagrass	South Atlantic, within state waters only

3.1 Analysis of Potential Routes of Effects Not Likely to Adversely Affect or Have No Effect on Listed Species or Designated Critical Habitat

After reviewing the proposed action, we believe the proper scope of the effects analysis for this Opinion is: 1) the effect that our exemption on the take of sea turtles through our sea turtle

[2] The South Atlantic (SA), Carolina, Chesapeake Bay, and New York Bight DPSs are listed as endangered, while the Gulf of Maine DPS is listed as threatened.

conservation regulations has on listed species; 2) the effect sea turtle conservation regulations have on listed species; and 3) the effect that the federally-authorized shrimp fisheries (also subject to the sea turtle conservation regulations) have on listed species. Since the purpose of the sea turtle conservation regulations is to conserve all sea turtles in both state and federal waters, and the TED regulations provide an exemption to various shrimp trawl fishers in state waters to incidentally capture sea turtles (e.g., bait shrimp, pusher-head trawl, and wing net vessels), we evaluate the regulations' sufficiency through this Opinion and the jeopardy standard. We also look at how the sea turtle conservation regulations may affect other species via our TED requirements and tow time restrictions. We have not promulgated any Section 4(d) rules applicable to the shrimp fisheries that exempt the take of any other species beside sea turtles. Therefore, we do not bear responsibility for the take of these other listed species in state-managed fisheries and do not authorize that take via the ITS in this Opinion. Finally, we evaluate the effects of our authorization of the federal shrimp fisheries via the aforementioned FMPs, where we are solely responsible for all of the effects on listed species.

Based on the above, we have determined that the proposed action is not likely to adversely affect any listed whales (i.e., sei, blue, fin, North Atlantic right, sperm, humpback, or Bryde's whales), shortnose sturgeon, Nassau grouper, oceanic whitetip or scalloped hammerhead shark, or corals, and would have no effect on Johnson seagrass. We have also determined that the proposed action is not likely to adversely affect designated critical habitats for Gulf sturgeon and corals, and will have no effect on designated critical habitats for North Atlantic right whale, smalltooth sawfish, and Johnson's seagrass. These species and critical habitats are excluded from further analysis and consideration in this Opinion. The following discussion summarizes our rationale for these determinations.

Whales

All species of listed large whales protected by the ESA, with the exception of Bryde's whale, may be found in or near the Atlantic portion of the action area. In the Gulf of Mexico portion of the action area, Bryde's and sperm whales are the only endemic populations of whales. Blue, fin, sei, and sperm whales are predominantly found seaward of the continental shelf in waters where most shrimping does not occur. Sightings of sperm whales are almost exclusively in the continental shelf edge and continental slope areas (Scott and Sadove 1997). Sei and blue whales also typically occur in deeper waters and neither is commonly observed in the waters of the Gulf of Mexico or off the U.S. East Coast (CeTAP 1982; Waring et al. 2006; Waring et al. 2002; Wenzel et al. 1988). Fin whales are generally found along the 100-m depth contour with sightings also spread over deeper water including canyons along the shelf break (Waring et al. 2006). North Atlantic right whales and humpback whales are coastal animals and sighted in the nearshore environment in the Atlantic along the southeastern United States from November through March. North Atlantic right and humpback whales have also been spotted in the Gulf of Mexico, but only very rarely, and these sightings are thought to be inexperienced juveniles.

The only potential route of effect from the proposed action on whales is via vessel collisions with federally-permitted vessels fishing in federal waters or entanglement in their nets. There have been no reported interactions between offshore or coastal large whales and trawls in the Atlantic or Gulf of Mexico (76 FR 73912). In the rare event that a listed whale is in the same vicinity of a shrimp trawl, shrimp trawlers move slowly (e.g., average 2007-2010 observed shrimp vessel speed for all areas and fisheries [i.e., Gulf of Mexico penaeid, South Atlantic penaeid or rock shrimp] was 2.8 km, in Scott-Denton et al. 2012). This would give a whale or the fishing vessel time to avoid a collision or entanglement.

Bryde's whales may be affected by the royal red shrimp component of the shrimp fishery because the area in which these trawls operate overlap with a portion of the Bryde's whale biologically important area. The Bryde's whale status review concluded that royal red shrimp trawls are unlikely to have an interaction with a Bryde's whale because the overlap between royal red shrimp trawling and the Bryde's whale biologically important area is limited, and the effort in those overlapping areas represent a small portion of fishing effort (Rosel et al. 2016). In addition, royal red shrimp trawls are slow moving, and although these trawls use over a mile of cable, the gear is very taut due to the depths fished. This reduces the likelihood of an entanglement interaction. Lastly, there are no known royal red shrimp trawl fishery entanglements.

In summary, based on the above information, we conclude the proposed action is extremely unlikely to adversely affect any large whale protected by the ESA.

Shortnose Sturgeon

Shortnose sturgeon can be found in a number of river systems near the Atlantic portion of the action area. The shortnose sturgeon is considered a freshwater amphidromous species in the northeastern United States, rather than an anadromous one (Kieffer and Kynard 1993). Although it may exhibit a slightly greater tendency to use saline habitats in the southern portion of its range, the shortnose sturgeon rarely occurs in coastal waters where the shrimp trawl fisheries are pursued (Collins et al. 1996). A shortnose sturgeon entering federal waters and being captured during shrimp trawling is extremely unlikely to occur. It is possible that there is a very small amount of overlap between state-managed trawl fisheries during winter months. However, in the rare event a shortnose sturgeon interacts with a shrimp trawl in state waters, our implementation of the sea turtle conservation regulations would be expected benefit shortnose sturgeon. The required use of TEDs in the shrimp otter trawls is likely to provide any shortnose sturgeon that enters the trawl with a route of escape. During TED testing conducted by the SEFSC, TEDs were estimated to exclude 87% of encountered sturgeon (i.e., Atlantic and Gulf sturgeon) from capture by trawl nets. Given both Gulf and Atlantic sturgeon use TEDs to escape capture in trawl nets, presumably shortnose sturgeon would also be able to escape. Also, the required tow time restrictions under the sea turtle conservation regulations for other types of trawls (e.g., skimmer trawls less than 40 ft in length) may also benefit shortnose sturgeon by reducing the amount of time a shortnose sturgeon would spend trapped in the net before detected and released. The exemption of sea turtle take via the sea turtle conservation regulations is expected to have no effect on shortnose sturgeon, because it is unrelated to fishery operations and the low number of listed species interactions generally. Therefore, our implementation of the sea turtle conservation regulations and the exemption of sea turtle take through those actions would either have no effect or a solely beneficial effect on how state-authorized trawling affects shortnose sturgeon.

Nassau Grouper

Nassau grouper's preferred habitat in federal waters (i.e., generally associated with high-relief coral reef or rocky substrate) is not compatible with shrimp habitat, which is typically sandy and muddy habitat. Further, the SEFSC Observer Program has never observed Nassau grouper bycatch in the shrimp fisheries (E. Scott-Denton, NMFS, pers. comm., October 7, 2020). Based on this information, we conclude the proposed action is extremely unlikely to adversely affect Nassau grouper.

Oceanic Whitetip Shark

The oceanic whitetip shark is a pelagic species generally found in the open ocean, close to the surface, in water depths greater than 600 ft. As such, we don't expect any interaction with the shrimp fisheries, which utilize bottom trawl gear, typically in shallow, coastal waters. Therefore, we conclude the proposed action will have no effect on oceanic whitetip sharks.

Corals

The proposed action is not likely to adversely affect listed corals. The only potential route of effect from the proposed action on listed corals is via physical damage from trawling in federal waters. However, adverse effects from the fishery on these corals are extremely unlikely to occur, given differences between shrimp and coral preferred habitats, and protective regulations in place prohibiting or limiting trawling in areas where corals are most likely to occur.

White shrimp appear to prefer muddy or peaty bottoms when in inshore waters and soft, muddy bottoms when offshore. Brown shrimp appear to prefer a similar bottom type and may also be found in areas of unconsolidated sediment (i.e., mud, sand, and shell). Pink shrimp are found most commonly on unconsolidated sediment (SAFMC 1996). Rock shrimp are targeted in waters 130-300 ft off the eastern Florida coast. Royal red shrimp occur only in the very deep waters (780-1,800 ft) of the South Atlantic and Gulf of Mexico EEZ. Listed *Acropora* corals are found in waters less than 100 ft and are considered to be environmentally sensitive, requiring relatively clear, well circulated waters with optimal water temperatures of 25°-29°C. Thus, shrimp habitats are extremely unlikely to support listed coral species. Within the action area, elkhorn and staghorn corals may both occur near the Florida Keys and off the east coast of Florida in waters less than 100 ft. The maximum northern extent of elkhorn and staghorn corals is Broward County and Palm Beach County, respectively. Only approximately 249 mi^2 of Gulf of Mexico EEZ waters around the Florida Keys are within the potential depth range of these species. A single colony of elkhorn coral has been observed in the Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. The other listed corals extend north to Martin County, Florida and to depths of approximately 330 ft in hard-bottom areas where light is not limited by water clarity. They occur in the Florida Garden Banks National Marine Sanctuary and other reefs in the Gulf of Mexico (e.g., Pulley's ridge). Like *Acropora* species, they require relatively clear, well-circulated waters and are unlikely to occur in shrimp habitat. Protective regulations are in place prohibit or limit trawling in these areas (i.e., East and West Flower Garden Banks, Tortugas Shrimp Sanctuary). Regulations at 15 CFR 922.164 provide additional protection for corals occurring within existing management areas inside the Florida Keys National Marine Sanctuary. Most applicable is that the use of bottom trawls and other bottom tending gears are prohibited in these areas.

In summary, based on the above information, we conclude the proposed action is extremely unlikely to adversely affect any coral species protected by the ESA.

Johnson's Seagrass
Johnson's seagrass grows only along approximately 124 mi of coastline in southeastern Florida north of Sebastian Inlet, Indian River County, south to Virginia Key in northern Biscayne Bay, Miami-Dade County. Within that area, Johnson seagrass occurs in a patchy, disjoined distribution from the intertidal zone to depths of approximately 6-10 ft in a wide range of sediment types, salinities, and in variable water quality conditions (NMFS 2007a). There is no overlap between Johnson seagrass and federally-permitted vessels in the shrimp trawl fisheries. Johnson seagrass in the action area is contained within shallow state waters. It is possible that there is a very small amount of overlap between Johnson seagrass and state-managed shrimp trawl fisheries. Potential effects to Johnson seagrass from state-authorized trawling stem from trawls being dragged over Johnson seagrass and potentially uprooting them. However, the proposed action in state waters is limited to implementation of the sea turtle conservation regulations and the exemption of sea turtle take through those actions, which would have no effect on how state-authorized trawling may affect Johnson seagrass. The proposed sea turtle conservation regulations are aimed at providing a way for mobile animals to escape from inside shrimp trawl nets and do not change the way the gear interacts with the seafloor. As a result, we conclude the proposed action would have no effect on Johnson's seagrass.

Loggerhead Sea Turtle NWA DPS Critical Habitat
On July 10, 2014, we designated critical habitat along the southeast Atlantic coast of the United States, around the Florida peninsula, and through the Gulf of Mexico to Texas for the NWA DPS of the loggerhead sea turtle (79 FR 39855). Loggerhead critical habitat is divided into 5 different units: nearshore reproductive habitat, winter habitat, breeding habitat, constricted migratory habitat, and *Sargassum* habitat. The nearshore reproductive habitat unit is located in nearshore waters extending out 1.6 km offshore; thus, this unit is located solely within state waters, it falls outside our action area. For the other units, we do not expect the proposed action would affect the primary constituent elements (i.e., water temperature and depth for wintering habitat; proximity to the primary Florida migratory corridor and nesting grounds for breeding habitat; constricted continental shelf area and passage conditions for migration to and from nesting, breeding, and/or foraging areas for constricted migratory habitat; and concentrated components of the *Sargassum* community in water temperatures and depths suitable for the optimal growth of *Sargassum* and inhabitance of loggerhead sea turtles). Therefore, we conclude the proposed action will have no effect on critical habitat for the NWA DPS of the loggerhead sea turtle.

North Atlantic Right Whale Critical Habitat
Designated North Atlantic right whale critical habitat (50 FR 28793) can be found in the Atlantic portion of the action area from the mouth of the Altamaha River, Georgia, to Jacksonville, Florida, out 15 nm and from Jacksonville, Florida, to Sebastian Inlet, Florida, out 5 nm. However, there are no potential routes of effect from the proposed action on North Atlantic right whale critical habitat. The proposed action will have no effect on the physical and biological features (water depth, water temperature, and the distribution of right whale cow/calf pairs and the distance from the shoreline to the 130-ft depth contour [Kraus et al. 1993]), which were the basis for determining this habitat to be critical. Shrimp trawling involves pulling gear through the water along the sea floor and does not result in any changes to the water depth or temperature of where the gear is fished. Right whale cow/calf pair sighting s are distributed from shore out to 130 ft, but the average water depth at of sighting was 41.3 ft (standard deviation [SD]= 23.3 ft). The average water depth that South Atlantic penaeid shrimp vessels fish in is 28.9 ft, thus in shallower waters, which is shallower than where most cow/calf pairings are sighted, and rock shrimp are fished for in much deeper waters (i.e., water depth average of approximately 200 ft). Therefore, we conclude the proposed action will have no effect on North Atlantic right whale critical habitat.

Gulf Sturgeon Critical Habitat
We, along with USFWS, jointly designated Gulf sturgeon critical habitat on April 18, 2003 (50 CFR 226.214). Fourteen areas (units) are designated as Gulf sturgeon critical habitats; of which seven occur in the action area: Unit 8 (Lake Pontchartrain [east of causeway], Lake Catherine, Little Lake, the Rigolets, Lake Borgne, Pascagoula Bay, and Mississippi Sound systems in Louisiana and Mississippi, and sections of the state waters within the Gulf of Mexico); Unit 9 (Pensacola Bay system in Florida); Unit 10 (Santa Rosa Sound in Florida); Unit 11 (Nearshore Gulf of Mexico in Florida); Unit 12 (Choctawhatchee Bay system in Florida); Unit 13 (Apalachicola Bay system in Gulf and Franklin Counties, Florida); and Unit 14 (Suwannee Sound in Florida). The physical and biological features identified as essential for the conservation of the Gulf sturgeon within these waters are abundant prey items; water and sediment quality necessary for normal behavior, growth, and viability of all life stages; and, safe unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats.

We believe the proposed action is not likely to adversely affect Gulf sturgeon critical habitat. The critical habitat units above are all contained within state waters where the proposed action is limited to implementation of the sea turtle conservation regulations and the exemption of sea turtle take. These proposed actions have no effect on the Gulf sturgeon essential features relating to prey items and water and sediment quality (i.e., they do not change the way trawls interact with the sea floor, and therefore, have no effect on the abundance of prey items or water and sediment quality). We expect the TED requirements to be solely beneficial by maintaining unobstructed migratory pathways via providing a mechanism for Gulf sturgeon to escape and continue on their path in the event that they captured by a shrimp trawl in state waters fishing under the authority of that state.

Smalltooth Sawfish Critical Habitat

On September 2, 2009, we issued a final rule (74 FR 45353; see also, 50 CFR 226.218) to designate critical habitat for the U.S. DPS of smalltooth sawfish. The critical habitat consists of two units: the Charlotte Harbor Estuary Unit, which comprises approximately 221,459 acres (346 mi^2) of coastal habitat, and the Ten Thousand Islands/Everglades Unit, which comprises approximately 619,013 acres (967 mi^2) of coastal habitat in southwest Florida. The critical habitat units are both contained within state waters. The key conservation objective for the critical habitat units is to facilitate recruitment into the adult population by protecting juvenile nursery areas. The essential features of smalltooth sawfish critical habitat are: 1) red mangroves; and 2) shallow, euryhaline (fluctuating salinity) habitats characterized by water depths between mean high water and 3 ft measured at mean lower low waterline.

Designated critical habitat for the U.S. DPS of smalltooth sawfish is contained within state waters where the proposed action is limited to implementation of sea turtle conservation regulations and the exemption of sea turtle take through those actions. The sea turtle conservation regulations provide a way for mobile animals to escape from inside shrimp trawl nets through TEDs or at the water's surface when operating under alternative tow times. TEDs and tow time limits do not alter red mangroves or red mangrove habitat, depth, or salinity. Therefore, we conclude the proposed action will have no effect on the essential features identified in the critical habitat designation for the U.S. DPS of smalltooth sawfish, and thus, no effect on designated critical habitat.

Elkhorn and Staghorn Coral Critical Habitat

We designated critical habitat for elkhorn and staghorn corals in a final rule published on November 26, 2008 (73 FR 72209). The only potential route of effect from the proposed action on *Acropora* designated critical habitat is physical damage from federally-permitted vessels trawling in federal waters. The proposed action will have no effect on critical habitat contained within state waters, because the proposed action in such areas is limited to implementation of the sea turtle conservation regulations and the exemption of sea turtle take. These actions would have no effect on the physical and biological features identified as essential for *Acropora* corals, because they lack any potential to affect substrate quality, sedimentation, or macroalgal coverage. Areas of critical habitat occurring in the federal waters portion of the action area are limited to a small portion of the South Atlantic. The feature essential to the conservation of *Acropora* species is substrate of suitable quality and availability (i.e., "natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover"), in water depths from the mean high water line to approximately 100 ft. Because of the habitat preferred by commercially exploited shrimp species (as discussed above in our analysis of the proposed action's effects to coral species), fishing targeting these species is unlikely to occur on hard substrate of suitable quality and availability for *Acropora* species. Thus, adverse effects from the fisheries on *Acropora* critical habitat are extremely unlikely to occur.

Johnson's Seagrass Critical Habitat
Johnson's seagrass critical habitat is designated to include substrate and water in the following ten portions of the Indian River Lagoon and Biscayne Bay, Florida, within the current range of Johnson's seagrass (See 50 CFR 226.213 for geographic coordinates): 1) North of Sebastian Inlet Channel; 2) South of Sebastian Inlet Channel, 3) Fort Pierce Inlet; 4) North of St. Lucie Inlet; 5) Hobe Sound; 6) South side of Jupiter Inlet; 7) a portion of Lake Worth Lagoon north of Bingham Island; 8) a portion of Lake Worth Lagoon, located just north of the Boynton Inlet; 9) a portion of northeast Lake Wyman, Boca Raton; and 10) a portion of Northern Biscayne Bay. The essential features of Johnson seagrass critical habitat are: 1) adequate water quality; 2) adequate salinity levels; 3) adequate water transparency; and 4) stable, unconsolidated sediments that are free from physical disturbance.

Johnson seagrass critical habitat areas are all contained within shallow state waters where the proposed action is limited to implementation of the sea turtle conservation regulations and the exemption of sea turtle take through those actions. These actions would have no effect on the physical and biological features identified as essential for Johnson's seagrass, because they lack any potential to affect water quality, water transparency, salinity, or unconsolidated sediments.

3.2 Potential Routes of Effects Likely to Adversely Affect Listed Species

We anticipate that Kemp's ridley, green, loggerhead, leatherback, and hawksbill sea turtles, as well as Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish may be adversely affected by the proposed action due to the potential for fisheries bycatch. A discussion on these effects is included in Section 5.

3.2.1 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. The threats 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 sea turtle species (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 and other 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 (i.e., hundreds of sea turtles as calculated in this Opinion) 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 captures 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 2020a). 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), 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 (DWH) 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 2016). 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). The potential effects, and the expected related effects to ESA-listed species, stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future

(e.g., several decades) is fraught with uncertainty. As previously mentioned, we have elected to view the effects of climate change on affected species on a more manageable and predictable 10-year time period due to this reality.

While we cannot currently predict impacts on sea turtles stemming from climate change with any degree of certainty, we are aware that 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 over time 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).

A combination of rising sea surface temperatures that could alter nesting behavior to more northern latitudes and sea level rise resulting in increased beach erosion north of Cape Hatteras, North Carolina (Sallenger et al. 2012) and reduced availability of existing beaches, could ultimately affect sea turtle nesting success in those areas. However, we expect those effects, should they occur, would likely occur over a fairly long time period encompassing several sea turtle generations, and not in the short term (e.g., over the next decade). Furthermore, modeled climate data from Van Houtan and Halley (2011) showed a future positive trend for loggerhead nesting in Florida, by far the species' most important nesting area in the Atlantic, with increases through 2040 as a result of the Atlantic Multidecadal Oscillation signal. A more recent study by Arendt et al. (2013), which is a follow up review and critique of the Van Houtan and Halley (2011) analysis, suggested the mechanistic underpinning between climate and loggerhead nesting rates on Florida beaches was primarily acting on the mature adult females as opposed to the hatchlings. Nonetheless, Arendt et al. (2013) suggest that the population of loggerheads nesting in Florida could attain the demographic criteria for recovery by 2027 if annual nest counts from 2013-2019 are comparable to what were seen from 2008-2012. Since loggerhead sea turtles are known to nest on Florida beaches in large numbers (and likely will continue to do so in the short-term future), we believe that any impacts of the sea level rise described in Sallenger et al. (2012) are likely to be offset by increased nesting in Florida over the next few decades.

Other changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen [DO] 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 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 1), 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 declined to 17,945, with another steep drop to 11,090 nests in 2019 (Gladys Porter Zoo 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 [NPS] 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, the record nesting in 2017, and then a drop back down to 190 nests in 2019 (NPS data).

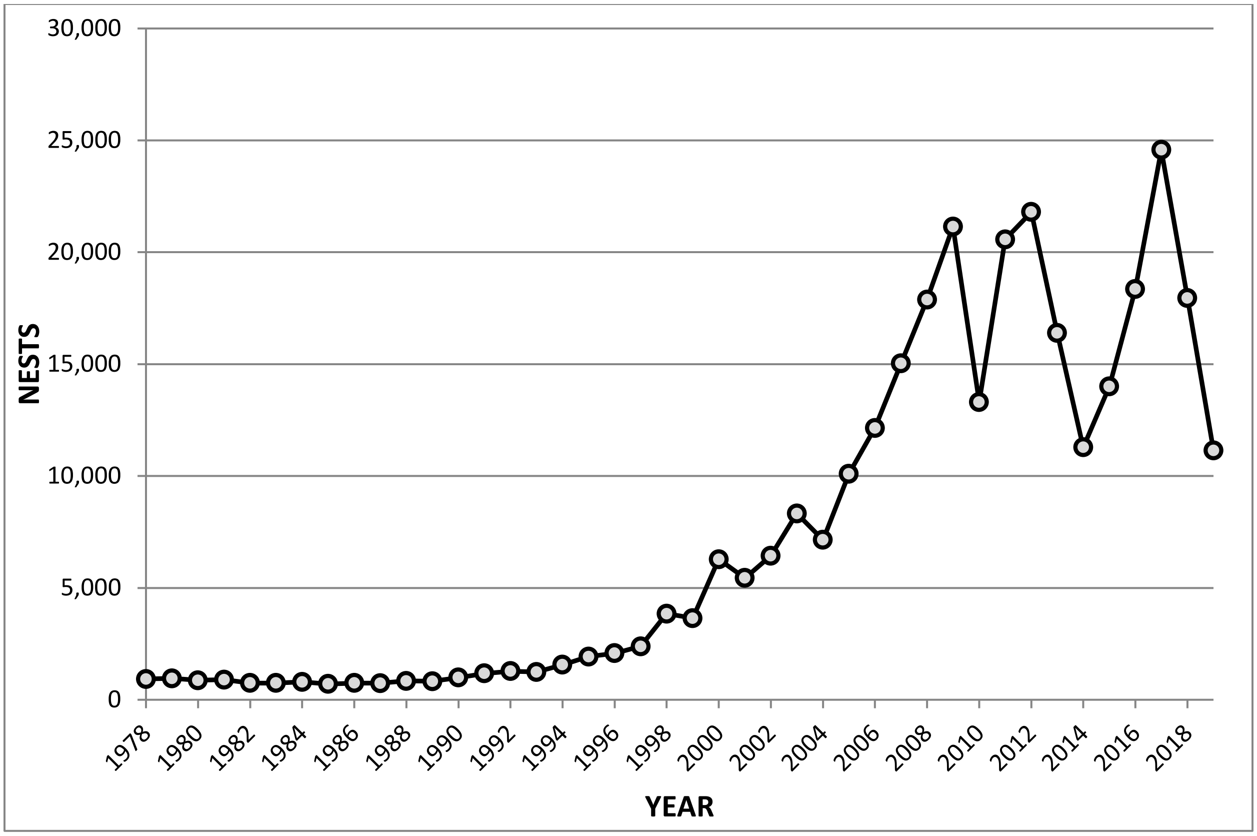

Figure 1. Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019).

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 3.2.1; 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*[3] 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 [STSSN] 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

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

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 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 beginning in 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). Subsequent years of observation noted additional captures in the skimmer trawl fisheries, including some mortalities. 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 4-in bar spacing TEDs in the skimmer trawl fisheries (77 FR 27411) was not implemented. Following additional gear testing, however, we proposed a new rule in 2016 (81 FR 91097) to require TEDs with 3-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 ft and greater in length to use TEDs designed to exclude small sea turtles in their nets effective April 1, 2021. As we previously noted, we delayed the effective date of this final rule until August 1, 2021, due to safety and travel restrictions related to the COVID-19 pandemic that prevented necessary training and outreach for fishers. 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 3.2.1, 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.

3.2.3 Green Sea Turtle

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 DPSs (81 FR 20057 2016) (Figure 2). 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 North Atlantic DPS (NA DPS) and South Atlantic DPS (SA 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.

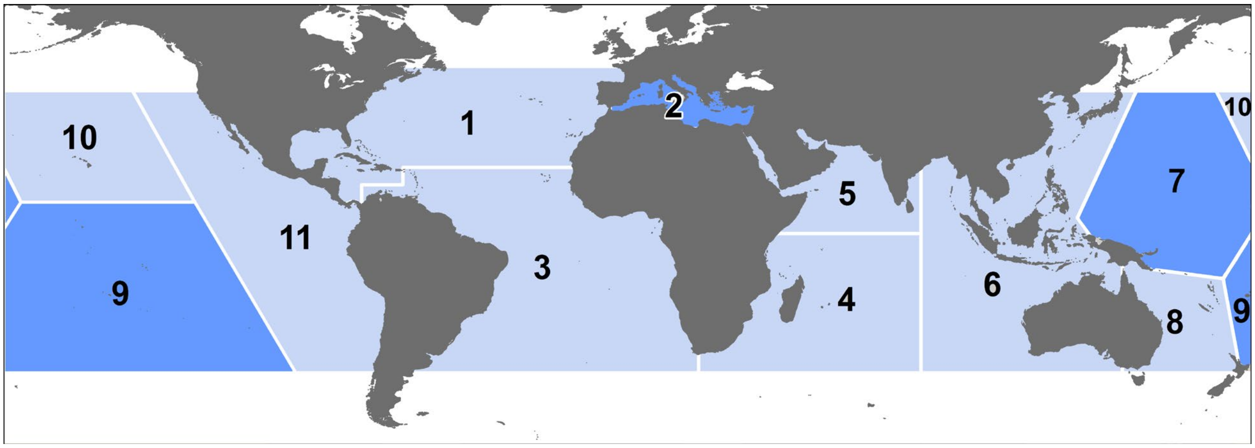

Figure 2. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic (NA); 2. Mediterranean; 3. South Atlantic (SA); 4. Southwest Indian; 5. North Indian; 6. East Indian-West Pacific; 7. Central West Pacific; 8. Southwest Pacific; 9. Central South Pacific; 10. Central North Pacific; 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 pounds (lb) (159 kilograms [kg]) with an SCL 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 deoxyribonucleic acid (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.

NA DPS Distribution

The NA DPS boundary is illustrated in Figure 2. 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.

SA DPS Distribution

The SA DPS boundary is shown in Figure 2, 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 (NA DPS) (Naro-Maciel et al. 2007; Naro-Maciel et al. 2012). While no nesting occurs as far south as Uruguay and Argentina, both have important foraging grounds for South Atlantic green turtles (Gonzalez Carman et al. 2011; Lezama 2009; López-Mendilaharsu et al. 2006; Prosdocimi et al. 2012; Rivas-Zinno 2012).

Life History Information

Green sea turtles reproduce sexually, and mating occurs in the waters off nesting beaches and along migratory routes. Mature females return to their natal beaches (i.e., the same beaches where they were born) to lay eggs (Balazs 1982; Frazer and Ehrhart 1985) every 2-4 years while males are known to reproduce every year (Balazs 1983). In the southeastern 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 in (5 cm) in length and weigh approximately 0.9 ounces (oz). 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 2007c). Green sea turtles exhibit particularly slow growth rates of about 0.4-2 in (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 in (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 2007c).

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.

<u>NA DPS Status and Population Dynamics</u>
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 2007c). 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 2007c). 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).

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 3). According to data collected from Florida's index nesting beach survey from 1989-2019, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 40,911 in 2019. 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). 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.

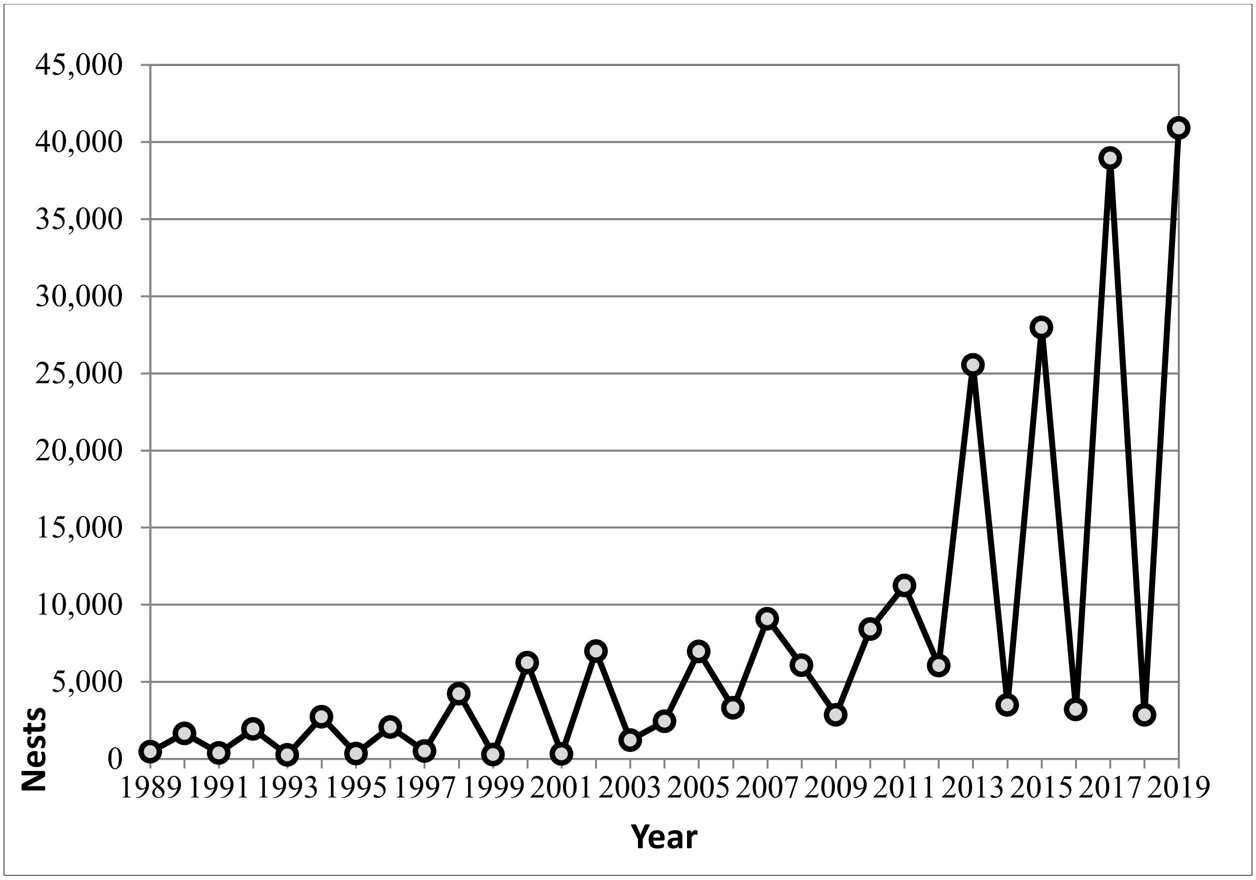

Figure 3. Green sea turtle nesting at Florida index beaches since 1989.

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

<u>SA DPS Status and Population Dynamics</u>

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 (United Kingdom), Aves Island (Venezuela), and Galibi (Suriname) appear to be increasing. Others such as Trindade (Brazil), Atol das Rocas (Brazil), and Poilão (Guinea-Bissau) 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 3.2.1.

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 in (0.1 cm) to greater than 11.81 in (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 3.2.1, 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 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources, which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

While green turtles regularly use the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic, and the proportion of the population using the northern Gulf of Mexico at any given time is relatively low. Although it is known that adverse impacts occurred and numbers of animals in the Gulf of Mexico were reduced as a result of the DWH oil spill of 2010, 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 2016).

3.2.4 Loggerhead Sea Turtle (NWA DPS)

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. We, along with USFWS, published a final rule on September 22, 2011, which designated 9 DPSs for loggerhead sea turtles (76 FR 58868, 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) 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 use 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 (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[4]); 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 in 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 in (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

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

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 Indian River Lagoon, Florida, are regularly used by juveniles but not by adult loggerheads. Adult loggerheads do tend to use estuarine areas with more open ocean access, such as the Chesapeake Bay in the U.S. mid-Atlantic. Shallow-water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads (Conant et al. 2009).

Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has also been documented (Hawkes et al. 2007; Georgia Department of Natural Resources [GADNR], unpublished data; South Carolina Department of Natural Resources [SCDNR], 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 2009a; 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.

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

In addition to the total nest count estimates, the Florida Fish and Wildlife Research Institute (FWRI) uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. This provides a better tool for understanding the nesting trends (Figure 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 nonsignificant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening confidence intervals. Nesting at the core index beaches declined in 2017 to 48,033, and rose slightly again to 48,983 in 2018 and then 53,507 in 2019, which is the 3rd 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).

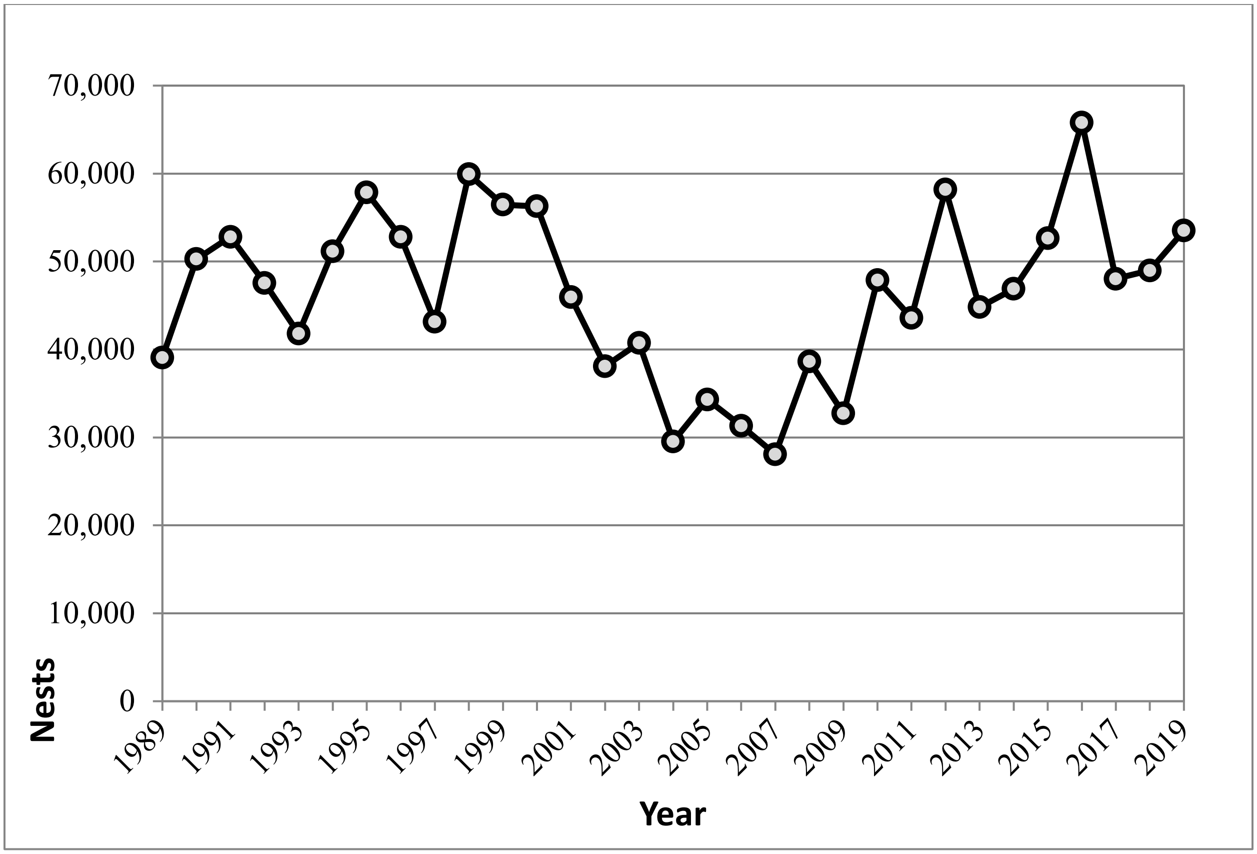

Figure 4. Loggerhead sea turtle nesting at Florida index beaches since 1989.

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 (GADNR unpublished data, North Carolina Wildlife Resources Commission [NCWRC] unpublished data, 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) 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. 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 5).

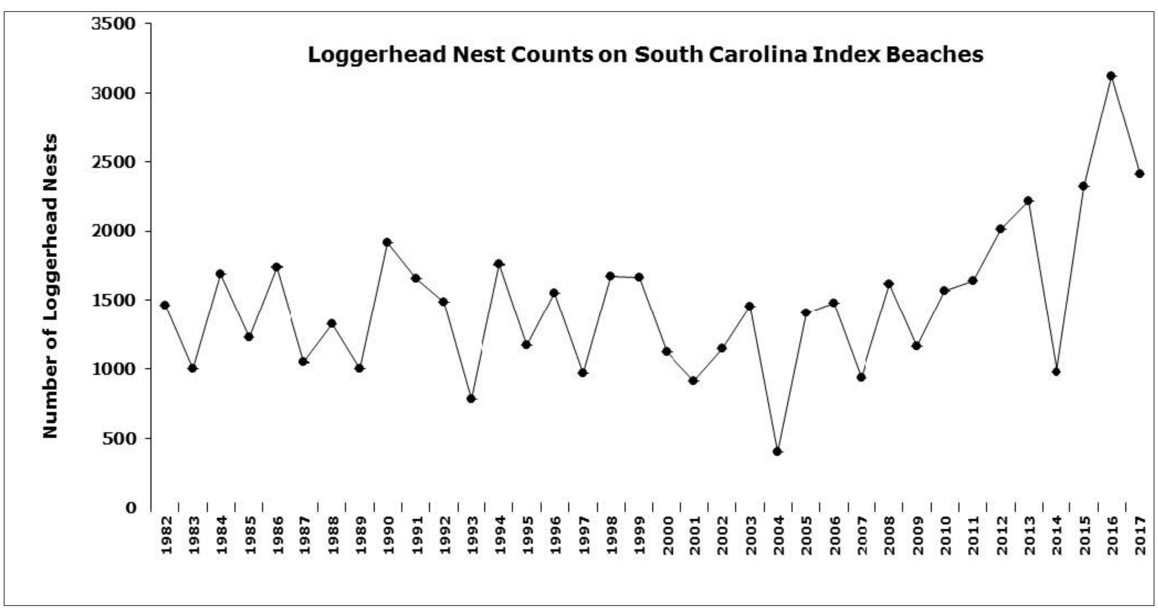

Figure 5. South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: http://www.dnr.sc.gov/seaturtle/nest.htm).

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

Our SEFSC developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS 2009a). 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 2009a). 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 2009a). 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 2011a).

Threats

The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 3.2.1. Yet the impact of fishery interactions is a point of further emphasis for this species. The joint 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 3.2.1, specific impacts of the DWH oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (DWH Trustees 2016). 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 that 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 NFMRU), the DWH Trustees (2016) 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). We expect these issues may affect other sea turtle species similarly.

3.2.5 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 CCL that often exceeds 5 ft (150 cm) and front flippers that can span almost 9 ft (270 cm) (NMFS and USFWS 1998a). 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),[5] a thick layer of insulating fat (Davenport et al. 1990; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),[6] 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 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

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[5] 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.

[6] "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.

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,[7] 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 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).

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

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 [FWC], 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 6 and Table 4). A similar pattern was also observed statewide (Table 4). 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 4. Number of Leatherback Sea Turtle Nests in Florida.

Nests Recorded	2011	2012	2013	2014	2015	2016	2017	2018	2019
Index Nesting Beaches	625	515	322	641	489	319	205	316	337
Statewide	1,653	1,712	896	1,604	1,493	1,054	663	949	1,090

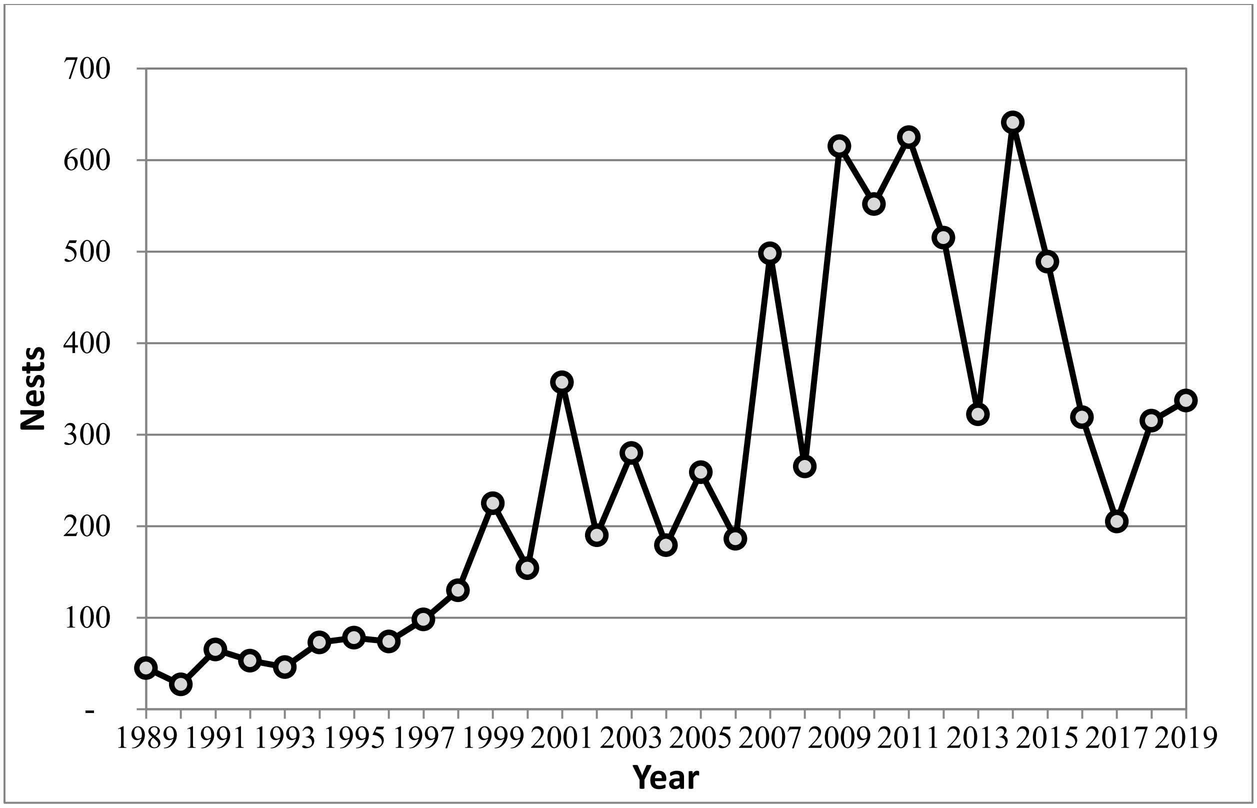

Figure 6. Leatherback sea turtle nesting at Florida index beaches since 1989.

The West African nesting stock of leatherback s 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). TEWG (2007) also determined that at the time of their publication, leatherback sea turtle populations in the Atlantic were all stable or increasing with the exception of the Western Caribbean and West Africa populations. A later review by NMFS and USFWS (2013a) suggested the leatherback nesting population was stable in most nesting regions of the Atlantic Ocean. However, as described earlier, the Northwest 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 Northwest 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 3.2.1; 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 that 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 3.2.1, 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 2007d). 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 3.2.1, 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. 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.

3.2.6 Hawksbill Sea Turtle

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

Species Description and Distribution
Hawksbill sea turtles are small- to medium-sized (99-150 lb on average [45-68 kg]) although females nesting in the Caribbean are known to weigh up to 176 lb (80 kg) (Pritchard et al. 1983).

The carapace is usually serrated and has a "tortoise-shell" coloring, ranging from dark to golden brown, with streaks of orange, red, and/or black. The plastron of a hawksbill turtle is typically yellow. The head is elongated and tapers to a point, with a beak-like mouth that gives the species its name. The shape of the mouth allows the hawksbill turtle to reach into holes and crevices of coral reefs to find sponges, their primary adult food source, and other invertebrates. The shells of hatchlings are 1.7 in (42 mm) long, are mostly brown, and are somewhat heart-shaped (Eckert 1995; Hillis and Mackay 1989; van Dam et al. 1990).

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 1998b; 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 2007e).

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.htmhttp://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 2007e). 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 2007e).

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 3.2.1. 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 3.2.1, 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 2016). 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.

3.2.7 Atlantic Sturgeon

We listed 5 separate DPSs of Atlantic sturgeon under the ESA on February 6, 2012 (77 FR 5880), which was effective on April 6, 2012. Specifically, we listed the New York Bight, Chesapeake Bay, Carolina, and SA DPSs as endangered, while we listed the Gulf of Maine DPS as threatened.

Species Description and Distribution
Atlantic sturgeon are long-lived, late-maturing, estuarine-dependent, anadromous fish distributed along the east coast of North America (Waldman and Wirgin 1998). Historically, sightings have been reported from Hamilton Inlet, Labrador, Canada, south to the St. Johns River, Florida (Murawski et al. 1977; Smith and Clugston 1997). Atlantic sturgeon may live up to 60 years, reach lengths up to 14 ft, and weigh over 800 lbs (ASSRT 2007; Collette and Klein-MacPhee 2002). They are distinguished by armor-like plates called scutes and a long protruding snout that has four barbels, which are slender, whisker-like feelers extending from the lower jaw used for touch and taste. Adult Atlantic sturgeon spend the majority of their lives in nearshore marine waters, returning to their natal rivers (i.e., rivers where they were born) to spawn (Wirgin et al. 2002). Young sturgeon may spend the first few years of life in their natal river estuary before moving out to sea (Wirgin et al. 2002). Atlantic sturgeon are omnivorous benthic (i.e., bottom) feeders and incidentally ingest mud along with their prey. Diets of adult and subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as

sand lance (ASSRT 2007; Bigelow and Schroeder 1953a; Guilbard et al. 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (ASSRT 2007; Bigelow and Schroeder 1953a; Guilbard et al. 2007).

Historic U.S. distribution of Atlantic sturgeon included approximately 38 rivers from the St. Croix River, Maine to the St. Johns River, Florida, of which 35 rivers have been confirmed to have had a historical spawning subpopulation. Presently, the SA DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) of the Ashepoo, Combahee, and Edisto Rivers' basin (ACE Basin) southward along the South Carolina, Georgia, and Florida coastal areas to the Saint Johns River, Florida. The Carolina DPS includes all Atlantic sturgeon that are spawned in the watersheds (including all rivers and tributaries) from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor. The Chesapeake Bay DPS is comprised of Atlantic sturgeon that originate from rivers that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, Virginia. The New York Bight DPS includes all anadromous Atlantic sturgeon that spawn in the watersheds that drain into coastal waters from Chatham, Massachusetts, to the Delaware-Maryland border on Fenwick Island. The Gulf of Maine DPS includes all anadromous Atlantic sturgeons that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, Massachusetts.

The marine range of all 5 DPS of Atlantic sturgeon extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida. The action area physically includes natal rivers of the South Atlantic and Carolina DPSs. The location of the action means subadult and adults could be affected by the action, however, because adult and subadult Atlantic sturgeon from all DPSs mix extensively in marine waters, we expect fish from all DPSs to potentially be found within the action area.

Life History Information
Atlantic sturgeon are generally referred to as having 4 size/developmental categories: larvae; young-of-year (YOY); juveniles and subadults; and adults. Hatching occurs approximately 94-140 hours after egg deposition. Immediately after hatching larvae enter the yolk sac larval stage and assume a demersal existence (Smith et al. 1980). The yolk sac provides nutrients to the animals until it is completely absorbed 8-12 days after hatching (Kynard and Horgan 2002). Animals in this stage are fewer than 4 weeks old, with total lengths (TL) less than 30 mm (Van Eenennaam et al. 1996a). Animals in this phase are in freshwater and are located far upstream very near the spawning beds. As the larvae develop they commence downstream migration towards the estuaries. During the first half of their downstream migration, movement is limited to night. During the day, larvae use gravel, rocks, sticks, etc., as refugia (Kynard and Horgan 2002). During the latter half of migration when larvae are more fully developed, movement occurs both day and night. Salinities of 5-10 parts per thousand are known to cause mortality at this young stage (Bain 1997; Cech and Doroshov 2005; Kynard and Horgan 2002). As larvae grow and absorb the yolk sac, they enter the YOY phase. YOY are greater than 4 weeks old but

less than 1 year, and generally occur in the natal river. These animals are generally located downstream of the spawning beds in primarily freshwater, though they can be found in the estuaries.

Following the YOY life phase, sturgeon develop into juveniles and subadults. There is little morphometric difference, aside from overall size, between juveniles and subadults; they are primarily distinguished by their occurrence within estuarine or marine waters. Juveniles are generally only found in estuarine habitats, while subadults may be found in estuarine and marine waters. As a group, juveniles and subadults range in size from approximately 300-1500 mm TL. The term "juveniles" refers to animals 1 year of age or older that reside in the natal estuary. Estuarine habitats are important for juveniles, serving as nursery areas by providing abundant foraging opportunities, as well as thermal and salinity refuges, for facilitating rapid growth. During their first 2 years, juvenile Atlantic sturgeon remain in the estuaries of their natal rivers, which may include both fresh and brackish channel habitats below the head of tide (Hatin et al. 2007). Upon reaching age 2, juveniles become increasingly salt tolerant and some individuals will begin their outmigration to nearshore marine waters (Bain 1997; Dovel and Berggren 1983; Hatin et al. 2007). Some juveniles will take up residency in non-natal rivers that lack active spawning sites (Bain 1997). By age 5, most juveniles have completed their transition to saltwater becoming "subadults," "late-stage juveniles," or "marine migratory juveniles," however, these animals are frequently encountered in estuaries of non-natal rivers (Bahr and Peterson 2016).

Out migration of larger juveniles may be influenced by the density of younger, less-developed juveniles. Because early juveniles are intolerant of salinity, they are likely unable to use foraging habitats in coastal waters if riverine food resources become limited. However, older, more-developed juveniles are able to use these coastal habitat, though they may prefer the relatively predator-free environments of brackish water estuaries as long as food resources are not limited (Schueller and Peterson 2010).

Adults are sexually mature individuals of 1500+ mm TL and 5 years of age or older. They may be found in freshwater riverine habitats on the spawning grounds or making migrations to and from the spawning grounds. They also use estuarine waters seasonally, principally in the spring through fall and will range widely in marine waters during the winter. After emigration from the natal estuary, subadults and adults travel within the marine environment, typically in waters shallower than 50 m in depth, using coastal bays, sounds, and ocean waters often occurring over sand and gravel substrate (Collins and Smith 1997; Dunton et al. 2010; Erickson et al. 2011; Greene et al. 2009)

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

Spawning adult Atlantic sturgeon generally migrate upriver in spring to early summer, which occurs in February-March in southern systems, April-May in Mid-Atlantic systems, and May-July in Canadian systems (Bain 1997; Caron et al. 2002; Murawski et al. 1977; Smith 1985; Smith and Clugston 1997). Likely fall spawning runs have been identified in the Edisto River, South Carolina (Farrae et al. 2017) and the Altamaha River, Georgia (Ingram and Peterson 2016). Telemetry data collected in 2013 and 2015 also show acoustically tagged fish making spawning runs in late summer (August-September) in the Savannah River (SCDNR, unpublished data). A fall spawning run has also been confirmed in the Roanoke River, North Carolina (Smith et al. 2015), in the Carolina DPS, however, they report a spring spawning run is also likely occurring. This suggests that a fall spawn is occurring in a number of southern rivers (Collins et al. 2000b; McCord et al. 2007; Moser et al. 1998; Rogers and Weber 1995; Weber and Jennings 1996). Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers, when and where optimal flows are 46-76 cm/sec and depths are 3-27 m (Bain et al. 2000; Borodin 1925; Crance 1987; Leland 1968; Scott and Crossman 1973). Males commence upstream migration to the spawning sites when waters reach around 6°C (Dovel and Berggren 1983; Smith 1985; Smith et al. 1982) with females following a few weeks later when water temperatures are closer to 12° or 13°C (Collins et al. 2000a; Dovel and Berggren 1983; Smith 1985). Atlantic sturgeon have highly adhesive eggs that must be laid on hard bottom in order to stick. Thus, spawning occurs over hard substrate, such as cobble, gravel, or boulders (Gilbert 1989; Smith and Clugston 1997).

Status and Population Dynamics

At the time Atlantic sturgeon were listed, the best available abundance information for each of the 5 DPSs was the estimated number of adult Atlantic sturgeon spawning in each of the rivers on an annual basis. However, the estimated number of annually spawning adults in each of the river subpopulations is insufficient to quantify the total population numbers for each DPS of Atlantic sturgeon due to the lack of other necessary accompanying life history data. In 2012, the NEFSC estimated the total ocean population of adults and subadults, vulnerable to capture in fisheries within the sampling domain of the Northeast Area Monitoring and Assessment Program (NEAMAP). NEAMAP trawl surveys were conducted from Cape Cod, Massachusetts, to Cape Hatteras, North Carolina, in nearshore waters to depths of 60 ft, from fall 2007 through spring 2012.

The results of these surveys are presented in Table 5. It is important to note that the NEAMAP surveys were conducted primarily in the Northeast and may underestimate the actual population abundances of the Carolina and SA DPSs, which are likely more concentrated in the Southeast since they originated from and spawn there. However, the total ocean population abundance estimates listed in Table 5 currently represent the best available population abundance estimates for the 5 U.S. Atlantic sturgeon DPSs.

Table 5. Summary of Calculated Population Estimates based upon the NEAMAP Survey Swept Area, Assuming 50% Efficiency (Damon-Randall et al. 2013; NMFS 2013).

DPS	Mean Percent Composition Estimate for Each DPS	Estimated Ocean Population Abundance	Estimated Ocean Population of Adults	Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries)
South Atlantic	20%	14,911	3,728	11,183
Carolina	4%	1,356	339	1,017
Chesapeake Bay	14%	8,811	2,203	6,608
New York Bight	49%	34,566	8,642	25,925
Gulf of Maine	11%	7,455	1,864	5,591

Since the 2012 listing, 2 mixed stock analyses have been completed for Atlantic sturgeon: 1 evaluating individuals captured in the Northeast United States (from north of Cape Hatteras to Maine) and 1 evaluating individuals in the Southeast United States (from south of Cape Hatteras to central east Florida). A mixed stock analysis (MSA) is currently the best available method for identifying which DPSs are most likely to be encountered in the marine environment. An MSA pools all the genetic information available for Atlantic sturgeon caught in a given area and evaluates the proportional representation of each river of origin and DPS within that catch area. The proportion of animals from a specific DPS or river of origin found in a given catch area, is directly related to the distance between the catch area and those rivers/DPSs. For example, the SA DPS includes rivers from Florida, Georgia, and South Carolina. Thus, for a catch area off the coast of Georgia, we anticipate a high proportion of individuals from the SA DPS and a low proportion from the Gulf of Maine DPS. Similarly, individuals from DPSs with larger populations are expected to occur at higher proportions overall than animals from DPSs with relatively smaller populations.

GARFO applied the results of the MSA specific to the Northeast Region (Damon-Randall et al. 2013) to their 2012 estimate (Table 5), to estimate the likely population of adults and subadults from each DPS captured in the NEAMAP trawl surveys conducted from Cape Cod, Massachusetts, to Cape Hatteras, North Carolina.

The U.S. Geological Survey (USGS) Leetown Science Center completed a draft MSA specific to the Southeast Region in late 2019 (USGS unpublished data). USGS provided information on both river of origin and DPS of origin (Table 6). Specifically, the report evaluated the genetic information from a given individual and determined which of 14 potential reference rivers it was most likely to have originated from. Individuals found in Southeast only assigned to 12 of those reference rivers; individuals from 2 rivers in Canada (St. John and St. Lawrence) were not detected, as shown in Table 6. USGS (unpublished data) used the same approach to assign individuals in the Southeast to a likely DPS of origin.

Prior to the completion of the Southeast-specific MSA, we used the population estimates (Table 5), and MSA for the Northeast (Damon-Randall et al. 2013), for projects occurring in the Southeast because they represented the best available information. This Southeast-specific MSA significantly improves the accuracy with which we can assign incidental bycatch occurring for federal actions taking place in the Southeast. However, with the new analysis, it is no longer appropriate to use the total ocean population estimates of adults and subadults based on data from the NEAMAP program, because those estimates were based on individuals collected outside the Southeast. Unfortunately, no estimates of the total populations in the Southeast have been completed. In the absence of a total population estimate, we relied on the best river-specific estimates available to develop an estimate of abundance for Atlantic sturgeon in the Southeast. We therefore calculated population estimates of the juvenile and adult populations of Atlantic sturgeon in the Southeast (described below) using demographic information from the Altamaha River combined with the proportions on individuals from specific rivers and DPSs provided in the draft 2019 MSA (USGS unpublished data).

We used the Altamaha River in the SA DPS as the foundation for our estimates because it has the most available information. The scientific literature provides estimates of Age 1, 2, and 3 abundances for the Altamaha River in 2004-2006 (Schueller and Peterson 2010), as well as estimates of the number of adults likely making spring spawning runs in 2004 and 2005 (Peterson et al. 2008). Ingraham and Peterson (2016) calculated the likely proportion of sexually mature adults in the Altamaha system that make spawning runs in the spring, allowing us to extrapolate the total number of spawning adults in the Altamaha River. We summed the estimates for all age classes (i.e., Age 1 juveniles to extrapolated total spawning adults) to estimate a likely total population of Atlantic sturgeon in the Altamaha River. Following this approach, we estimated the minimum total juvenile and adult spawning population in the Altamaha River was between 1,940 and 2,525 individuals (Table 6).

Once we estimated the likely total population for the Altamaha River for 2004 and 2005, we used the information in the draft 2019 MSA (USGS unpublished data) to calculate the likely number of individuals from other rivers/DPS that likely occur in the Southeast. Specifically, since we estimated the minimum total juvenile and adult spawning population in the Altamaha River was between 1,940 and 2,525 individuals, and the MSA estimated the Altamaha River accounted from approximately 20.2% of the individuals in the Southeast, we estimated the total Southeast population of Atlantic sturgeon as between 9,583-12,477 (5,867-27,387). Table 6 outlines this approach and provides our estimates.

The resulting estimates are conservative and likely represent a minimum numbers of animals because they do not account for YOY individuals, as YOY individuals are so small they are difficult to capture and were not a focus of the Altamaha River sampling. Likewise, they only estimate the likely population present in the Southeast (e.g., south of Cape Hatteras to central east Florida). This is significant because while individuals from the northern DPSs (Gulf of Maine, New York Bight, and Chesapeake Bay DPSs) could be in the action area, those animals are likely transients. Thus, our estimates of individuals from those DPSs occurring in the Southeast is unlikely to accurately reflect the total population abundance for the Gulf of Maine, New York Bight, and Chesapeake Bay DPSs. Additionally, some portion of the South Atlantic and Carolina DPSs are likely to travel north of Cape Hatteras. We discuss available information and population estimates by DPS below.

Table 6. Estimated Atlantic Sturgeon Population in the Southeast.[8]

STEP 1				
Year	Age 1 (Min/Max)	Age 2 (Min/Max)	Age 3 (Min/Max)	Total (Min/Max)
2004	483 (368-643)	544 (424-707)	37 (9-294)	1,064 (801-1,644)
2005	1345 (1,077-1,697)	107 (28-784)	30 (6-935)	1,482 (1,111-3,416)
STEP 2				
Year	Estimated Spring Spawning Run (Min/Max)	Estimated Proportion of Adults Making Spring Spawning Run (Min/Max)	Total Adults (Min/Max)	
2004	324 (143-667)	0.37 (0.36-0.38)	876 (386-1,802)	
2005	386 (216-787)	0.37 (0.36-0.38)	1,043 (584-2,217)	
STEP 3				
Year	Total (Min/Max)			
2004	1,940 (1,187-3,447)			
2005	2,525 (1,695-5,542)			
STEP 4				
River Population (DPS)	Proportion of Individuals from River Population (CI)	DPS	Proportion of Individuals from DPS (CI)	
Kennebec (Gulf of Maine)	0.001 (0-0.004)	Gulf of Maine	0.001 (0-0.004)	
Hudson (New York Bight)	0.025 (0.015-0.035)	New York Bight	0.036 (0.025-0.048)	
Delaware (New York Bight)	0.011 (0.004-0.0200	Chesapeake	0.096 (0.079-0.121)	
York (Chesapeake Bay)	0.004 (0.001-0.008)	Carolina	0.338 (0.292-0.364)	
James (Chesapeake Bay)	0.093 (0.076-0.116)	South Atlantic	0.529 (0.499-0.570)	
Albemarle Complex (Carolina)	0.309 (0.259-0.331)			
Pee Dee (Carolina)	0.030 (0.011-0.053)			
Edisto Spring (South Atlantic)	0.020 (0.012-0.034)			
Edisto Fall (South Atlantic)	0.100 (0.081-0.124)			

[8] The table follows these 5 steps: 1) sum juvenile abundance in Altamaha River (Schueller and Peterson 2010); 2) estimate adult population in Altamaha River (Ingram and Peterson 2016; Peterson et al. 2008a); 3) sum age class estimates in Altamaha River; 4) identify the proportion of individual river systems/DPS represented in the Southeast (USGS unpublished data); and 5) estimate individuals from remaining river populations and DPSs, using the Altamaha estimates.

Savannah (South Atlantic)	0.102 (0.070-0.137)			
Ogeechee (South Atlantic)	0.070 (0.054-0.098)			
Altamaha (South Atlantic)	0.202 (0.171-0.238)			
Satilla (South Atlantic)	0.036 (0.020-0.049)			
STEP 5				
River Population (DPS)	Proportion of Individuals in SE by River Population (CI)	Minimum Number of Individuals in SE by River Populations (Min/Max)	Proportion of Individuals from DPS	Minimum Number of Individuals in SE By DPS
Kennebec (Gulf of Maine)	0.001 (0-0.004)	9 (5-25)	Gulf of Maine 0.001 (0-0.004)	9
Hudson (New York Bight)	0.025 (0.015-0.035)	241 (147-687)	New York Bight 0.036 (0.025-0.048)	343
Delaware (New York Bight)	0.011 (0.004-0.0200	103 (63-293)	Chesapeake 0.096 (0.079-0.121)	920
York (Chesapeake Bay)	0.004 (0.001-0.008)	35 (21-99)	Carolina 0.338 (0.292-0.364)	3,243
James (Chesapeake Bay)	0.093 (0.076-0.116)	886 (543-2,533)	South Atlantic 0.529 (0.499-0.570)	5,067
Albemarle Complex (Carolina)	0.309 (0.259-0.331)	2,957 (1,810-8,451)		
Pee Dee (Carolina)	0.030 (0.011-0.053)	286 (175-816)		
Edisto Spring (South Atlantic)	0.020 (0.012-0.034)	188 (115-537)		
Edisto Fall (South Atlantic)	0.100 (0.081-0.124)	954 (584-2,725)		
Savannah (South Atlantic)	0.102 (0.070-0.137)	973 (596-2,780)		
Ogeechee (South Atlantic)	0.070 (0.054-0.098)	666 (408-1,903)		
Altamaha (South Atlantic)	0.202 (0.171-0.238)	1,940 (1,187-5,543)		
Satilla (South Atlantic)	0.036 (0.020-0.049)	348 (213-994)		

SA DPS

Secor (2002) estimates that 8,000 and 11,000 adult females were likely present in South Carolina and Georgia respectively prior to 1890. The sturgeon fishery was the third largest fishery in Georgia until the fishery's collapse in the late 1800s.

The SA DPS historically supported 8 spawning subpopulations. At the time of listing only 6 spawning subpopulations were believed to have existed: the Combahee, Edisto, Savannah, Ogeechee, Altamaha (including the Oconee and Ocmulgee tributaries), and Satilla Rivers. We determined those rivers/river systems supported spawning if YOY were observed, or mature adults were present, in freshwater portions of a system. Three of the spawning subpopulations in the SA DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. These spawning subpopulations are likely less than 6% of their historical abundance. There are an estimated 343 adults that spawn annually in the Altamaha River and fewer than 300 adults spawning annually (total of both sexes) in the river systems where spawning still occurs (75 FR 61904; October 6, 2010). The abundance of the remaining 3 spawning subpopulations in the SA DPS is likely less than 1% of their historical abundance (ASSRT 2007). The 2 remaining historical spawning subpopulations in the Broad-Coosawatchie and St. Marys Rivers were previously believed to be extinct. However, new information provided from the capture of juvenile Atlantic sturgeon suggests the spawning subpopulation in the St. Marys River is not extinct and continues to exist, albeit at very low levels. Regardless of river, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development.

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

The assessment also estimated effective population sizes when possible. Effective population size is generally considered to be the number of individuals that contribute offspring to the next generation. More specifically, based on genetic differences between animals in a given year, or over a given period of time, researchers can estimate the number of adults needed to produce that level of genetic diversity.

For the SA DPS, the assessment reported population size for the Edisto, Savannah, Ogeechee, and Altamaha Rivers (Table 7). Additional estimates of population size have been conducted since the completion of the assessment, including for additional river systems; Table 7 reports those estimates.[9]

Table 7. Estimates of Effective Population Size by Rivers.

River	Effective Population Size (95% CI)	Sample Size	Collection Years	Reference
Edisto	55.4 (36.8-90.6)	109	1996-2005	ASMFC (2017)
	Fall Run: 48.0 (44.7-51.5)	1,154	1996-2004	Farrae et al. (2017a)
	Spring Run: 13.3 (12.1-14.6)	198	1998, 2003	Farrae et al. (2017a)
	60.0 (51.9-69.0)	145	1996, 1998, 2005	Waldman et al. (2018)

[9] The effective population size estimates in Table 7 are different from the estimated population estimated Atlantic sturgeon population in the Southeast reported in Table 6. The effective population size estimates refer to the likely number of unique spawning individuals needed to produce the level of genetic variability seen in the population. The effective population size only considers spawning individuals and does not account for any other age classes. Therefore, an estimate of effective population size is *not* an estimate of population abundance and can be significantly lower than the total number of individuals when account for all age classes.

River	Effective Population Size (95% CI)	Sample Size	Collection Years	Reference
Savannah	126.5 (88.1-205)	98	2000-2013	ASMFC (2017)
	123 (103.1-149.4)	161	2013, 2014, 2017	Waldman et al. (2018)
Ogeechee	32.2 (26.9-38.8)	115	2003-2015	ASMFC (2017)
	26 23.9–28.2	200	2007-2009, 2014-2017	Waldman et al. (2018)
	23.9 (22.2-25.7)	197	2007-2009, 2014-2017	Fox et al. (2019)
Altamaha	111.9 (67.5-216.3)	186	2005-2015	ASMFC (2017)
	149 (128.7–174.3)	245	2005, 2011, 2014, 2016-2017	Waldman et al. (2018)
	142.1 (124.2-164.0)	268	2005, 2011, 2014-2017	Fox et al. (2019)
Satilla	21 (18.7–23.2)	68	2015-2016	Waldman et al. (2018)
St. Marys	1 (1.3–2.0)	14	2014-2015	Waldman et al. (2018)

Generally, a minimum population size of 100 individuals is considered the threshold required to limit the loss in total fitness from inbreeding depression to <10%; while a population size greater than 1,000 is the recommended minimum to maintain evolutionary potential (ASMFC 2017; Frankham et al. 2014). Population size is useful for defining abundance levels where populations are at risk of loss of genetic fitness (ASMFC 2017). While not inclusive of all the spawning rivers in the SA DPS, the estimates reported in Table 7 suggest there is a risk for inbreeding depression in 4 of those rivers (Edisto, Ogeechee, Satilla, and St. Marys rivers) and loss of evolutionary potential in all 6 rivers. This information suggests there at least some inbreeding depression within the DPS and loss of evolutionary potential throughout all of it.

The GARFO NEAMAP model estimates a minimum ocean population for the entire SA DPS of 14,911 Atlantic sturgeon, of which 3,728 are adults. The SERO estimate, based on the 2019 USGS MSA, is that the minimum number of individuals from the SA DPS occurring in the Southeast is 5,067.

Carolina DPS

Historical fishery landings data indicate between 7,000 and 10,500 adult female Atlantic sturgeon were present in North Carolina prior to 1890 (Armstrong and Hightower 2002; Secor 2002). Secor (2002) estimates that 8,000 adult females were present in South Carolina during that same time frame. At the time of listing, the abundance for each river population within the DPS was estimated to have fewer than 300 spawning adults; estimated to be less than 3% of what they were historically (ASSRT 2007).

We have identified 7 rivers/river systems within the Carolina DPS where spawning is likely occurring: 1) Roanoke; 2) Tar-Pamlico; 3) Neuse; 4) Cape Fear and Northeast Cape Fear; 5) Pee Dee, Waccamaw, and Bull Creek; 6) Black; and 7) Santee and Cooper. We determined those rivers/river systems supported spawning if YOY were observed, or mature adults were present, in freshwater portions of a system. However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development.

Historically, both the Sampit and Ashley Rivers in South Carolina were documented to have spawning subpopulations at one time. Yet, the spawning subpopulation in the Sampit River is believed to be extirpated and the current status of the spawning subpopulation in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning subpopulations.

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

For the Carolina DPS, the assessment only reported population size for the Albemarle Sound. Based on samples collected from 37 individuals from 1998-2008, the assessment estimated a population size of 14.2 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the Carolina DPS, this estimate suggests there is a risk for both inbreeding depression and loss of evolutionary potential in the DPS, assuming Albemarle Sound is representative of the entire DPS. The GARFO NEAMAP model estimates a minimum ocean population for the entire Carolina DPS of 1,356 Atlantic sturgeon, of which 339 are adult. We estimate the minimum number of individuals from the Carolina DPS occurring in the Southeast is 3,243.

Chesapeake Bay DPS

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

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

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

At the time of listing, the James River was the only known spawning river for the Chesapeake Bay DPS and spawning was believed to occur only in the spring, from approximately April-May, based on historical and current evidence (ASSRT 2007). Subsequently, new information for when and where spawning-condition adults were captured and tracked in the river led to the conclusion that Atlantic sturgeon spawn in the James River in both the spring and in the late summer to early fall (Balazik et al. 2012a; Balazik and Musick 2015). The results of genetic analyses support that the adults are 2 separate spawning groups. The genetic analyses also informed the effective population size of each group which were similar (Fall: 46 [95% CI: 32±71], Spring: 44 [95% CI: 26±79]) despite differences in the number of adults captured from each spawning subpopulation. From 2007 to 2016, 507 individual fall run Atlantic sturgeon were captured during the fall spawning and 40 individual Atlantic sturgeon were captured during the spring spawning (Balazik et al. 2017). This is a minimum count of the number of adult Atlantic sturgeon in the James River during the time period because capture efforts did not occur in all areas and at all times when Atlantic sturgeon were present in the river. There are no other estimates of abundance or trends in abundance for the James River spawning subpopulations.

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

The assessment reported population size for the York and James Rivers in the Chesapeake Bay DPS. In the York River, samples from 136 individuals collected from 2013-2015 produced an estimated population size of 7.8 individuals, while in the James River, 346 samples were collected from 1998-2015 and produced an estimated population size of 40.9 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the Chesapeake Bay DPS, these estimates at least hint that there is a risk for both inbreeding depression and loss of evolutionary potential in the DPS. The GARFO NEAMAP model estimates a minimum ocean population for the entire DPS of 8,811 Atlantic sturgeon, of which 2,319 are adults. We estimate the minimum number of individuals from the Chesapeake Bay DPS occurring in the Southeast is 920. Given the distance between the rivers of this DPS and Southeast, we anticipate these individuals would be sub-adults or adults.

<u>New York Bight DPS</u>

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

Prior to the onset of expanded fisheries exploitation of sturgeon in the 1800s, a conservative historical estimate for the Hudson River Atlantic sturgeon population was 10,000 adult females (Secor 2002). Current population abundance is likely at least one order of magnitude smaller than historical levels (ASSRT 2007; Kahnle et al. 2007; Secor 2002). Based on data collected from 1985-1995, an estimate of the mean annual number of mature adults (863 total: 596 males and 267 females) was calculated for the Hudson River riverine population (Kahnle et al. 2007). Kahnle (2007; 1998) also showed that the level of fishing mortality from the Hudson River Atlantic sturgeon fishery during the period of 1985-1995 exceeded the estimated sustainable level of fishing mortality for the riverine population, and may have led to reduced recruitment. At the time of listing, available data on abundance of juvenile Atlantic sturgeon in the Hudson River Estuary indicated a substantial drop in production of young since the mid-1970s (Kahnle et al. 1998). A decline appeared to occur in the mid- to late-1970s followed by a secondary drop in the late 1980s (ASMFC 2011; Kahnle et al. 1998; Sweka et al. 2007). CPUE data suggest that recruitment has remained depressed relative to catches of juvenile Atlantic sturgeon in the estuary during the mid- to late 1980s (ASMFC 2011; Sweka et al. 2007). From 1985-2007, there were significant fluctuations in CPUE. The number of juveniles appears to have declined between the late 1980s and early 1990s. While CPUE is generally higher in the 2000s as compared to the 1990s, significant annual fluctuations make it difficult to discern any trend. The CPUEs from 2000-2007 are generally higher than those from 1990-1999; however, they remain lower than the CPUEs observed in the late 1980s. Standardized mean catch per net set from the New York State Department of Environmental Conservation Juvenile Atlantic Sturgeon Survey have had a general increasing trend from 2006-2015, with the exception of a dip in 2013. There is currently not enough information regarding any life stage to establish a trend for the Hudson River population (ASMFC 2011; Sweka et al. 2007).

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

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

The assessment reported population size for the Hudson and Delaware Rivers in the New York Bight DPS. In the Hudson River, samples from 337 individuals collected from 1996-2015 produced an estimated population size of 144.2 individuals, while in the Delaware River, 181 samples were collected from 2009-2015 and produced an estimated population size of 56.7 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the New York Bight DPS, the estimates for the Hudson River suggests that spawning subpopulation may be large enough to avoid inbreeding depression. The Delaware River spawning subpopulation may still be at risk of inbreeding depression and both spawning subpopulations are likely at risk losing evolutionary potential. The GARFO NEAMAP model estimates a minimum ocean population for the entire DPS of 34,566 Atlantic sturgeon, of which 8,642 are adults. We estimate the minimum number of individuals from the New York Bight DPS occurring in the Southeast is 343. Given the significant distance between the rivers of this DPS and Southeast, we anticipate these individuals would be adults.

<u>Gulf of Maine DPS</u>

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

Historically, the Gulf of Maine DPS likely supported more than 10,000 spawning adults (ASSRT 2007; KRRMP 1993; Secor 2002). Other than the NEAMAP based estimates presented above, there are no empirical abundance estimates for the Gulf of Maine DPS. The ASSRT (2007) presumed that the Gulf of Maine DPS was comprised of fewer than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over two time periods, 1977-1981 and 1998-

2000, resulted in the capture of 9 adult and several hundred subadult Atlantic sturgeon (Squiers 2004). However, since the surveys were primarily directed at capture of smaller shortnose sturgeon, the gear used may not have been selective for larger, adult Atlantic sturgeon.

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

The assessment reported population size for the St. Lawrence, St. John, and Kennebec Rivers in the Gulf of Maine DPS. In the St. Lawrence, samples from 30 individuals collected in 2013 produced an estimated population size of 39 individuals; in the St. John River, 31 samples were collected from 1991-1993 and produced an estimated population size of 115 individuals; and for the Kennebec River, samples from 52 individuals were collected from 1980-2011, and produced an estimated population size of 63.4 individuals (ASMFC 2017). While not inclusive of all the spawning rivers in the Gulf of Maine DPS, the effective population size estimate for the St. John River suggests that spawning subpopulation may be large enough to avoid inbreeding depression. The estimates for the other 2 rivers, however, suggests those spawning subpopulations may be at risk, and all 3 spawning subpopulations are likely at risk losing evolutionary potential. The GARFO NEAMAP model estimates a minimum ocean population for the entire DPS of 7,455 Atlantic sturgeon, of which 1,864 are adults. We estimate the minimum number of individuals from the Gulf of Maine DPS occurring in the Southeast is 9 fish, and given the significant distance between the rivers of this DPS and Southeast, we anticipate these individuals would be adults.

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

Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry 1971; Shaffer

1981; Soulé 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life span allows multiple opportunities to contribute to future generations, it also increases the time frame over which exposure to the multitude of threats facing Atlantic sturgeon can occur.

The viability of the Atlantic sturgeon DPSs depends on having multiple self-sustaining riverine spawning subpopulations and maintaining suitable habitat to support the various life functions (spawning, feeding, growth) of Atlantic sturgeon populations. Because a DPS is a group of populations, the stability, viability, and persistence of individual populations affects the persistence and viability of the larger DPS. The loss of any population within a DPS will result in: 1) a long-term gap in the range of the DPS that is unlikely to be recolonized; 2) loss of reproducing individuals; 3) loss of genetic biodiversity; 4) potential loss of unique haplotypes; 5) potential loss of adaptive traits; 6) reduction in total number; and 7) potential for loss of population source of recruits. The loss of a population will negatively impact the persistence and viability of the DPS as a whole, as fewer than 2 individuals per generation spawn outside their natal rivers (King et al. 2001; Waldman et al. 2002; Wirgin et al. 2000). The persistence of individual populations, and in turn the DPS, depends on successful spawning and rearing within the freshwater habitat, the immigration into marine habitats to grow, and then the return of adults to natal rivers to spawn.

Threats

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

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

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

Within in the range of the SA DPS on the Savannah River, the New Savannah Bluff Lock and Dam at the City of Augusta, is located just a few kilometers below impassible rapids, denying Atlantic sturgeon access to 7% of its historically available habitat (ASSRT 1998). However, the Augusta Shoals, the only rocky shoal habitat on the Savannah River and the former primary spawning habitat for Atlantic sturgeon in the river (Duncan et al. 2003; Marcy et al. 2005; USFWS 2003; Wrona et al. 2007), is located above this dam, and is currently inaccessible to Atlantic sturgeon. So, while Atlantic sturgeon have access to the majority of historical habitat in terms of unimpeded river miles, only a small amount of spawning habitat exists downstream of this dam and the vast majority of the rocky freshwater spawning habitat they need is inaccessible as a result of the dam.

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

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

In the SA DPS, maintenance dredging is currently modifying Atlantic sturgeon nursery habitat in the Savannah River. Modeling indicates that the proposed deepening of the navigation channel will result in reduced DO and upriver movement of the salt wedge, restricting spawning habitat.

Dredging is also modifying nursery and foraging habitat in the St. Johns River. For the Carolina DPS, dredging in spawning and nursery grounds modifies the quality of the habitat and is further restricting the extent of available habitat in the Cape Fear and Cooper Rivers, where Atlantic sturgeon habitat has already been modified and restricted by the presence of dams. Dredging for navigational purposes is suspected of having reduced available spawning habitat for the Chesapeake Bay DPS in the James River (ASSRT 2007; Bushnoe et al. 2005; Holton and Walsh 1995). Both the Hudson and Delaware rivers have navigation channels that are maintained by dredging. Dredging is also used to maintain channels in the nearshore marine environment. Many rivers in the range of the Gulf of Maine DPS, including the Kennebec River, also have navigation channels that are maintained by dredging. Dredging outside of federal channels and in-water construction occurs throughout the range of the Chesapeake Bay, New York Bight and Gulf of Maine DPSs.

Dickerson (2013) summarized observed takings of 23 Atlantic sturgeon from dredging activities conducted by U.S. Army Corps of Engineers (USACE) and observed from 1990-2013. Of the 3 types of dredges considered by Dickerson (2013) (hopper, clamshell, and pipeline), most sturgeon were captured by hopper dredge, though some captures were also noted in clamshell and pipeline dredges. Notably, reports include only those trips when an observer was on board to document capture. Additional data provided by USACE indicate another 16 Atlantic sturgeon were killed by hopper dredging from 2016-2018 in the Southeast. To offset the adverse effects associated dredging relocation trawling is used at times. The USACE has used this technique during dredging at Brunswick Harbor, Savannah Harbor, Kings Bay, and in the Savannah River channel. Trawling in these area captured and relocated 215 Atlantic sturgeon from 2016-2018.

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

Reductions in water quality from terrestrial activities have modified habitat utilized by the SA DPS. Low DO is modifying sturgeon habitat in the Savannah due to dredging, and non-point source inputs are causing low DO in the Ogeechee River and in the St. Marys River, which completely eliminates juvenile nursery habitat in summer. Low DO has also been observed in the St. Johns River in the summer. In the Pamlico and Neuse systems of the Carolina DPS, nutrient-loading and seasonal anoxia are occurring, associated in part with concentrated animal feeding operations. Heavy industrial development and concentrated animal feeding operations have degraded water quality in the Cape Fear River. Water quality in the Waccamaw and Yadkin-Pee Dee Rivers has been affected by industrialization and riverine sediment samples contain high levels of various toxins, including dioxins.

Decreased water quality also threatens Atlantic sturgeon of the Chesapeake Bay DPS, especially since the Chesapeake Bay system is vulnerable to the effects of nutrient enrichment due to a relatively low tidal exchange and flushing rate, large surface-to-volume ratio, and strong stratification during the spring and summer months (ASMFC 1998; ASSRT 2007; Pyzik et al. 2004). These conditions contribute to reductions in DO levels throughout the bay. The availability of nursery habitat, in particular, may be limited given the recurrent hypoxia (low DO) conditions within the Bay (Niklitschek and Secor 2005; Niklitschek and Secor 2010).

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

Atlantic sturgeon may also be particularly susceptible to impacts from environmental contamination because they are long-lived, benthic feeders. Sturgeon feeding in estuarine habitats near urbanized areas may be exposed to numerous suites of contaminants within the substrate. Contaminants, including toxic metals, polycyclic aromatic hydrocarbons (PAHs), organophosphate and organochlorine pesticides, PCBs, and other chlorinated hydrocarbon compounds can have substantial deleterious effects on aquatic life. These elements and compounds can cause acute lesions, growth retardation, and reproductive impairment in fishes (ASSRT 2007; Cooper 1989; Sindermann 1994).

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

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

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

Large-scale factors impacting riverine water quality and quantity that likely exacerbate habitat threats to Atlantic sturgeon of all 5 DPSs include drought, and intra- and inter-state water allocation. Changes in the climate are very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. For example, annual precipitation in the Southeast has increased by 0.19 in per decade since 1950 (NCDC 2019) but has also experienced several significant periods of drought (i.e., categorized as “abnormally dry” to “exceptional”) since 2000 (NDMC 2018). The Northeast has seen even more significant increases in annual precipitation with increases of 0.71 in per decade since 1950 (NCDC 2019). While not as severe, the Northeast has also experienced 2 periods of notable drought since 2000, as well as multiple other dry periods during that period. Abnormally low stream flows can restrict access by sturgeon to habitat areas and exacerbate water quality issues such as water temperature, reduced DO, nutrient levels, and contaminants.

Long-term observations also confirm changes in temperature are occurring at a rapid rate. From 1895-2017, the average annual temperature in the Southeast and Northeast has risen 0.1°F and 0.2°F per decade, respectively . From 1950-2017, the increase triples to 0.3°F per decade per decade for both regions (NCDC 2019). Aside from observation, climate modeling also projects future increases in temperatures in both the Southeast and Northeast. Table 8 summarizes the projected increases by the mid-century (2036-2065) and late-century (2071-2100). These are projections based on 2 different Representative Concentration Pathway model scenarios used by the Intergovernmental Panel on Climate Change (IPCC), relative to average from 1976-2005 (Hayhoe et al. 2017).

Table 8. Projected Temperature Increase in the Southeast and Northeast Under 2 Representative Concentration Pathway Model Projections (RCP4.5 and RCP8.5) and Time Series (Mid-Century, 2036-2065 and Late Century, 2071-2100) (Hayhoe et al. 2017).

National Climate Assessment Region	RCP4.5 (2036–2065)	RCP8.5 (2036–2065)	RCP4.5 (2071–2100)	RCP8.5 (2071–2100)
Northeast	3.98°F	5.09°F	5.27°F	9.11°F
Southeast	3.40°F	4.30°F	4.43°F	7.72°F

Ocean temperature in the U.S. Northeast Shelf and surrounding Northwest Atlantic Ocean has increased faster than the global average over the last decade (Pershing et al. 2015), and other projections suggest this region will warm 2-3 times faster than the global average (Saba et al. 2016). A first-of-its-kind climate vulnerability assessment, conducted on 82 fish and invertebrate species in the U.S. Northeast Shelf, concluded that Atlantic sturgeon from all 5 DPSs were among the most vulnerable species to global climate change (Hare et al. 2016).

Sea-level rise is another consequence of climate change; it has already had significant impacts on coastal areas and these impacts are likely to increase. Since 1852, when the first topographic maps of the southeastern U.S. were prepared, high tidal flood elevations have increased approximately 12 in. During the 20th century, global sea level has increased 15 to 20 cm (NAST 2000). Sea level rise is also projected to extend areas of salinization of groundwater and estuaries. Some of the most populated areas of this region are low-lying; the threat of saltwater entering into this region's aquifers with projected sea level rise is a concern (USGRG 2004). Saltwater intrusion will likely exacerbate existing water allocation issues, leading to an increase in reliance on interbasin water transfers to meet municipal water needs, further stressing water quality. Similarly, saltwater intrusion is likely to affect local ecosystems. Analysts attribute the forest decline in the Southeast to saltwater intrusion associated with sea level rise. Coastal forest losses will be even more severe if sea level rise accelerates as is expected as a result of global warming.

The effects of future climate change will vary greatly in diverse coastal regions for the United States. Warming is very likely to continue in the United States during the next 25-50 years, regardless of reduction in greenhouse gases, due to emissions that have already occurred (NAST 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25-50 years, and it is possible that they will accelerate. A warmer and drier climate would reduce stream flows and increase water temperatures. Expected consequences would be a decrease in the amount of DO in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer, wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch et al. 2000).

Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer et al. 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change.

In marine waters, there is a high confidence that observed changes are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007).

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

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

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

Very little is known about the effects of vessel strikes on individuals from the Carolina or SA DPSs. However, there is increasing evidence that vessels may pose as significant a threat to Atlantic sturgeon in the southern portion of their range as it does further north. We do not have a dedicated sturgeon carcass/stranding program, so we rely on the public to report encounters. In mid-2018, we deployed signs in North Carolina asking the public to report sturgeon sightings (alive or dead) to gather more information. From 2018 through August 2019, we received 5 reports of dead Atlantic sturgeon in the Cape Fear River, North Carolina, that were likely struck by ships. Prior to the deployment of these signs, there were 2 reports of potential ship strikes in the Cape Fear River from 2011 to 2014. It is unclear if this uptick represents an increasing threat from vessels, or just increasing reports. The lower estuaries of rivers in the Carolina and SA DPSs are often marsh habitats that can be very difficult for the public to access. Given the geology of these rivers, it is possible, if not likely, that a significant number of sturgeon are being struck by vessels in the rivers of the Carolina and SA DPSs, but are not reported.

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to Atlantic sturgeon in all 5 DPSs. Atlantic sturgeon are more sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum reproductive rates, and a large percentage of egg production occurs later in life. Based on these life history traits, Boreman (1997) calculated that Atlantic sturgeon can only withstand the annual loss of up to 5% of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0% and 51%, with the greatest mortality occurring in sturgeon caught by sink gillnets. Currently, there are estimates of the number of Atlantic sturgeon captured and killed in sink gillnet and otter trawl fisheries authorized by FMPs in the Northeast Region (Miller and Shepherd 2011). Those estimates indicate from 2006-2010, on average there were 1,548 and 1,569 encounters per year in observed gillnet and trawl fisheries, respectively, with an average of 3,118 encounters combined annually. Mortality rates in gillnet gear were approximately 20%, while mortality rates in otter trawl gear are generally lower, at approximately 5%. Atlantic sturgeon are particularly vulnerable to being caught in sink gillnets; therefore, fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch. Atlantic sturgeon are incidentally captured in state and federal fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (ASMFC 2007; Stein et al. 2004). Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

Stochastic Events
Stochastic events, such as hurricanes, are common throughout the range of Atlantic sturgeon from all 5 DPSs. These events are unpredictable and their effect on the survival and recovery of the species in unknown; however, they have the potential to impede the survival and recovery directly if animals die as a result of them, or indirectly if habitat, is damaged as a result of these disturbances. For example, in 2018, flooding from Hurricane Florence flushed significant amounts of organic matter into rivers supporting Atlantic sturgeon in the Southeast. The DO levels in those rivers dropped so low that thousands of fish suffocated, including multiple sturgeon.

3.2.8 Gulf Sturgeon

Gulf sturgeon (*Acipenser oxyrinchus desotoi*) were listed as threatened effective October 30, 1991 (56 CFR 49653, September 30, 1991), after their stocks were greatly reduced or extirpated throughout much of their historic range by overfishing, dam construction, and habitat degradation. We jointly manage Gulf sturgeon with USFWS. In riverine habitats, USFWS is responsible for all consultations regarding Gulf sturgeon and critical habitat. In estuarine habitats, responsibility is divided based on the action agency involved. USFWS consults with the Department of Transportation, the Environmental Protection Agency, the U.S. Coast Guard (USCG), and the Federal Emergency Management Agency; we consult with the Department of Defense, USACE, the Bureau of Ocean Energy Management, and any other federal agencies not specifically mentioned at 50 CFR 226.214. In marine areas, we are responsible for all consultations regarding Gulf sturgeon and critical habitat. In 2009, we conducted a 5-year review with USFWS and found Gulf sturgeon continued to meet the definition of a threatened species (USFWS and NMFS 2009).

Species Description and Distribution
The Gulf sturgeon is a subspecies of the Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). Gulf sturgeon are nearly cylindrical fish with an extended snout, vertical mouth, 5 rows of scutes (bony plates surrounding the body), 4 chin barbels (slender, whisker-like feelers extending from the head used for touch and taste), and a heterocercal (upper lobe is longer than lower) caudal fin (tail fin). Adults range from 6-8 ft in length and weigh up to 200 lbs; females grow larger than males. Gulf sturgeon spawn in freshwater and then migrate to feed and grow in estuarine/marine (brackish/salt) waters. Large subadults and adults feed primarily on lancelets, brachiopods, amphipods and other crustaceans, polychaetes, and gastropods. Small Gulf sturgeons feed on benthic infauna such as amphipods, grass shrimp, isopods, oligochaetes, polychaetes, and chironomid and ceratopogonid larvae, found in the intertidal zone. Subadults of more than 5 kg and adults in the freshwater middle river reaches essentially fast during the summer and fall (Mason Jr. and Clugston 1993).

Historically, Gulf sturgeon occurred from the Mississippi River east to Tampa Bay. Sporadic occurrences were recorded as far west as the Rio Grande River in Texas and Mexico, and as far east and south as Florida Bay (Reynolds 1993; Wooley and Crateau 1985). The subspecies' present range extends from Lake Pontchartrain and the Pearl River system in Louisiana and Mississippi respectively, east to the Suwannee River in Florida.

Life History
Gulf sturgeon are long-lived, with some individuals reaching at least 42 years in age (Huff 1975). Age at sexual maturity ranges from 8-17 years for females and 7-21 years for males (Huff 1975). Chapman and Carr (1995) estimated that mature female Gulf sturgeon that weigh between 64 and 112 lb (29-51 kg) produce an average of 400,000 eggs. Spawning intervals range from 1-5 years for males, while females require longer intervals ranging from 3-5 years (Fox et al. 2000; Huff 1975).

Gulf sturgeon move from the Gulf of Mexico into coastal rivers in early spring (i.e., March through May). Fox et al. (2000) found water temperatures at time of river entry differed significantly by reproductive stage and sex. Individuals entered the river system when water temperatures ranged anywhere between 11.2°-27.1°C. Spawning occurs in the upper reaches of rivers in the spring when water temperature is around 15°-20°C. While Sulak and Clugston (1999) suggest that sturgeon spawning activity is related to moon phase, other researchers have found little evidence of spawning associated with lunar cycles (Fox et al. 2000; Slack et al. 1999). Fertilization is external; females deposit their eggs on the river bottom and males fertilize them. Gulf sturgeon eggs are demersal, adhesive, and vary in color from gray to brown to black (Huff 1975; Vladykov and Greely 1963). Parauka et al. (1991) reported that hatching time for artificially spawned Gulf sturgeon ranged from 85.5 hours at 18.4°C to 54.4 hours at about 23°C. Published research on the life history of younger Gulf sturgeon is limited. After hatching, YOY individuals generally disperse downstream of spawning sites, though some may travel upstream as well (Clugston et al. 1995; Sulak and Clugston 1999), and move into estuarine feeding areas for the winter months.

Tagging studies confirm that Gulf sturgeon exhibit a high degree of river fidelity (Carr 1983). Of 4,100 fish tagged, 21% (860 of 4,100 fish) were later recaptured in the river of their initial collection, 8 fish (0.2%) moved between river systems, and the remaining fish (78.8%) have not yet been recaptured (USFWS and GSMFC 1995). There is no information documenting the presence of spawning adults in non-natal rivers. However, there is some evidence of movements by both male and female Gulf sturgeon (n = 22) from natal rivers into non-natal rivers (Carr et al. 1996; Craft et al. 2001; Fox et al. 2002; Ross et al. 2001; Wooley and Crateau 1985).

Gene flow is low in Gulf sturgeon stocks, with each stock exchanging less than one mature female per generation (Waldman and Wirgin 1998). Genetic studies confirm that Gulf sturgeon exhibit river-specific fidelity. Stabile et al. (1996) analyzed tissue taken from Gulf sturgeon in 8 drainages along the Gulf of Mexico for genetic diversity and noted significant differences among Gulf sturgeon stocks, which suggests region-specific affinities and likely river-specific fidelity. Five regional or river-specific stocks (from west to east) have been identified: 1) Lake Pontchartrain and Pearl River; 2) Pascagoula River; 3) Escambia and Yellow Rivers; 4) Choctawhatchee River; and 5) Apalachicola, Ochlockonee, and Suwannee Rivers (Stabile et al. 1996).

After spawning, Gulf sturgeon move downstream to areas referred to as “summer resting” or “holding” areas. Adults and subadults are not distributed uniformly throughout the river, but instead show a preference for these discrete holding areas usually located in the lower and middle river reaches (Hightower et al. 2002). While it was suggested these holding areas were sought for cooler water temperatures (Carr et al. 1996; Chapman and Carr 1995), Hightower et al. (2002) found that water temperatures in holding areas where Gulf sturgeon were repeatedly found in the Choctawhatchee River were similar to temperatures where sturgeon were only occasionally found elsewhere in the river.

In the fall, movement from the rivers into the estuaries and associated bays begins in September (at water temperatures around 23°C) and continues through November (Foster and Clugston 1997; Huff 1975; Wooley and Crateau 1985). Because the adult and large subadult sturgeon have spent at least 6 months fasting or foraging sparingly on detritus (Mason Jr. and Clugston 1993) in the rivers, it is presumed they immediately begin foraging. Telemetry data indicate Gulf sturgeon are found in high concentrations near the mouths of their natal rivers with individual fish traveling relatively quickly between foraging areas where they spend an extended period of time (Edwards et al. 2007; Edwards et al. 2003).

Most subadult and adult Gulf sturgeon spend the cool winter months (October/November through March/ April) in the bays, estuaries, and the nearshore Gulf of Mexico (Clugston et al. 1995; Fox et al. 2002; Odenkirk 1989). Tagged fish have been located in well-oxygenated shallow water (less than 7 m) areas that support burrowing macro invertebrates (Craft et al. 2001; Fox and Hightower 1998; Fox et al. 2002; Parauka et al. 2001; Rogillio et al. 2007; Ross et al. 2001; Ross et al. 2009). These areas may include shallow shoals 5-7 ft (1.5-2.1 m), deep holes near passes (Craft et al. 2001), unvegetated sand habitats such as sandbars, and intertidal and subtidal energy zones (Abele and Kim 1986; Menzel 1971; Ross et al. 2009). Subadult and adult Gulf sturgeon overwintering in Choctawhatchee Bay, Florida, were generally found to occupy the sandy shoreline habitat at depths of 4-6 ft (2-3 m) (Fox et al. 2002; Parauka et al. 2001). These shifting, predominantly sandy, areas support a variety of potential prey items including estuarine crustaceans, small bivalve mollusks, ghost shrimp, small crabs, various polychaete worms, and lancelets (Abele and Kim 1986; Menzel 1971; Williams et al. 1989). Preference for sandy habitat is supported by studies in other areas that have correlated Gulf sturgeon presence to sandy substrate (Fox et al. 2002).

Gulf sturgeon are described as opportunistic and indiscriminate benthivores that change their diets and foraging areas during different life stages. Their guts generally contain benthic marine invertebrates including amphiopods, lancelets, polychaetes, gastropods, shrimp, isopods, molluscs, and crustaceans (Carr et al. 1996; Fox et al. 2002; Huff 1975; Mason Jr. and Clugston 1993). Generally, Gulf sturgeon prey are burrowing species that feed on detritus and/or suspended particles, and inhabit sandy substrate. In the river, YOY sturgeon eat aquatic invertebrates and detritus (Mason Jr. and Clugston 1993; Sulak and Clugston 1999) and juveniles forage throughout the river on aquatic insects (e.g., mayflies and caddisflies), worms (oligochaete), and bivalves (Huff 1975; Mason Jr. and Clugston 1993). Adults forage sparingly in freshwater and depend almost entirely on estuarine and marine prey for their growth (Gu et al. 2001). Both adult and subadult Gulf sturgeon are known to lose up to 30% of their total body weight while in fresh water, and subsequently compensate the loss during winter feeding in marine areas (Carr 1983; Clugston et al. 1995; Heise et al. 1999; Morrow et al. 1998; Ross et al. 2000; Sulak and Clugston 1999; Wooley and Crateau 1985).

Status and Population Dynamics

Abundance of Gulf sturgeon is measured at the riverine scale. Currently, 7 rivers are known to support reproducing populations of Gulf sturgeon: Pearl, Pascagoula, Escambia, Yellow, Choctawhatchee, Apalachicola, and Suwannee. Gulf sturgeon abundance estimates by river and year for the 7 known reproducing populations are presented in Table 9. The number of individuals within each riverine population is variable across their range, but generally over the last decade (USFWS and NMFS 2009) populations in the eastern part of the range (Suwannee, Apalachicola Choctawhatchee) appear to be relatively stable in number or have a slightly increasing population trend. In the western portion of the range, populations in the Pearl and Pascagoula Rivers, have never been nearly as abundant as those to the east, and their current status, post-hurricanes Katrina and Rita, is unknown as comprehensive surveys have not occurred.

Table 9. Gulf Sturgeon Abundance Estimates by River and Year, with Confidence Intervals (CI) for the 7 Known Reproducing Populations. Data from USFWS and NMFS (2009).

River	Year of data collection	Abundance Estimate	Lower Bound 95% CI	Upper Bound 95% CI	Source
Suwannee	2007	14,000	not reported	not reported	Sulak 2008
Apalachicola	1991	144	83	205	Zehfuss et al. 1999
Choctawhatchee	2008	3314	not reported	not reported	USFWS 2009
Yellow	2003 fall	911	550	1,550	Berg et al. 2007
Escambia	2006	451	338	656	USFWS 2007
Pascagoula	2000	216	124	429	Ross et al. 2001
Pearl	2001	430	323	605	Rogillio et al. 2001

Both acute and episodic events are known to impact individual populations of Gulf sturgeon that in turn, affect overall population numbers. For example, on August 9, 2011, an overflow of "black liquor" (an extremely alkaline waste byproduct of the paper industry) was accidentally released by a paper mill into the Pearl River near Bogalusa, Louisiana, that may have affected the status and abundance of the Pearl River population. While paper mills regularly use acid to balance the black liquor's pH before releasing the material, as permitted by the Louisiana Department of Environmental Quality, this material released was not treated.[10] The untreated waste byproduct created a low oxygen (i.e., hypoxic) environment lethal to aquatic life. These hypoxic conditions moved downstream of the release site killing fish and mussels in the Pearl River over several days. Within a week after the spill, the DO concentrations returned to normal in all areas of the Pearl River tested by Louisiana Department of Wildlife and Fisheries (LDWF). The investigation of fish mortality began on August 13, 2011, several days after the spill occurred. Twenty-eight Gulf sturgeon carcasses (38-168 cm TL) were collected in the Pearl River after the spill (Sanzenbach 2011a; Sanzenbach 2011b) and anecdotal information suggests many other Gulf sturgeon carcasses were not collected. The smaller fish collected represent YOY and indicate spawning is likely occurring in the Pearl River. The spill occurred during the time when Gulf sturgeon were still occupying the freshwater habitat. Because the materials moved downriver after the spill, the entire Pearl River population of Gulf sturgeon was likely impacted.

Threats

The 1991 listing rule (56 FR 49653) for Gulf sturgeon cited the following impacts and threats: 1) Dams on the Pearl, Alabama, and Apalachicola Rivers; also on the North Bay arm of St. Andrew Bay; 2) channel improvement and maintenance activities: dredging and de-snagging; 3) water quality degradation, and 4) contaminants. In 2009, we conducted a 5-year review of the Gulf sturgeon with USFWS and identified several new threats to the Gulf sturgeon (USFWS and NMFS 2009). The following is a comprehensive list of threats to Gulf sturgeon, additional details can be found in the 5-year status review (USFWS and NMFS 2009):

[10] The extreme alkalinity of the untreated black liquor caused it to quickly bond with oxygen (aerobic) to dissociate in water. This reduced the amount of oxygen available within the water column, creating a hypoxic environment (< 1mg/L of DO) lethal to aquatic life.

1) Pollution from industrial, agricultural, and municipal activities is believed responsible for a suite of physical, behavioral, and physiological impacts to sturgeon worldwide. Specific impacts of pollution and contamination on sturgeon have been identified to include muscle atrophy; abnormality of gonad, sperm, and egg development; morphogenesis of organs, tumors; and disruption of hormone production.

2) Chemicals and metals such as chlordane, dichlorodiphenyldichloroethylene, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later incorporated into the food web as they are consumed by benthic feeders, such as sturgeon or macroinvertebrates.

3) Bycatch from fisheries may continue although all directed fisheries of Gulf sturgeon have been closed since 1990 (USFWS and GSMFC 1995). Although confirmed reports are rare, it is a common opinion among Gulf sturgeon researchers that bycatch mortality continues.

4) Dredging activities can pose significant impacts to aquatic ecosystems by: a) direct removal/burial of organisms; b) turbidity/siltation effects; c) contaminant resuspension; d) noise/disturbance; e) alterations to hydrodynamic regime and physical habitat; and f) loss of riparian habitat. Dredging operations may also destroy benthic feeding areas, disrupt spawning migrations, and resuspend fine sediments causing siltation over required substrate in spawning habitat. Because Gulf sturgeon are benthic omnivores, the modification of the benthos affects the quality, quantity, and availability of prey.

5) Collisions between jumping Gulf sturgeon and fast-moving boats on the Suwannee River and elsewhere are a relatively recent and new source of sturgeon mortality and pose a serious public safety issue as well. The Florida Fish and Wildlife Commission documented 3 collisions in the Suwannee River in 2008, and 1 incident in 2009.

6) Dams represent a significant impact to Gulf sturgeon by blocking passage to historical spawning habitats, which reduces the amount of available spawning habitat or entirely impede access to it. The ongoing operations of these dams also affect downstream habitat.

7) Global climate change may affect Gulf sturgeon by leading to accelerated changes in habitats utilized by Gulf sturgeon through saltwater intrusion, changes in water temperature, and extreme weather periods that could increase both droughts and floods.

8) Hurricanes have resulted in mortality of Gulf sturgeon in both Escambia Bay after Hurricane Ivan in 2004 (USFWS 2005) and Hurricane Katrina in 2005.

9) Red tide is the common name for a harmful algal bloom (HAB) of marine algae (*Karenia brevis*) that produces a brevetoxin that is absorbed directly across the gill

membranes of fish or through ingestion of algal cells. Fish mortalities associated with *Karenia brevis* events are very common and widespread. Blooms of red tides have been increasing in frequency in the Gulf of Mexico since the 1990s and have likely killed Gulf sturgeon at both the juvenile and adult life stages.

10) Aquaculture: Although the state of Florida has Best Management Practices to reduce the risk of hybridization and escapement, the threat of introduction of captive fishes into the wild continues.

Summary of the Status of Gulf Sturgeon

In summary, the Gulf sturgeon population is estimated to number approximately 19,000 individuals. The number of individuals within each riverine population is variable across their range, but populations in the eastern part of the range (Suwannee, Apalachicola Choctawhatchee) appear to be relatively stable in number or have a slightly increasing population trend (Sulak et al. 2016). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Gulf sturgeon. Their late age at maturity provides more opportunities for individuals to be removed from the population before reproducing. While a long life span also allows multiple opportunities to contribute to future generations, this is hampered within the species' range by habitat alteration, pollution, and bycatch.

A wide range of threats continue to dictate the status of Gulf sturgeon and their recovery. Modification of habitat through dams, the operation of dams, and dredging particularly impact Gulf sturgeon. The presence of dams reduces the amount of available spawning habitat or entirely impedes access to it, while ongoing operation of these dams affects downstream water quality parameters such as depth, temperature, velocity, and DO. Similarly, dredging projects modify Gulf sturgeon spawning and nursery habitat through direct removal of habitat features or reduced water quality due to nutrient-loading, anoxia, and contaminated sediments. Water quality can be further influenced by inter-basin water transfers and climate change, which may exacerbate existing water quality issues. Further, access to habitat and water quality continues to be a problem even with our authority under the Federal Power Act to prescribe fish passage and existing controls on some pollution sources. The inadequacy of regulatory mechanisms to control habitat alterations is contributing to the status of Gulf sturgeon.

Bycatch is also a current threat to the species that is contributing to its status. Although confirmed reports are rare, it is a common opinion among Gulf sturgeon researchers that bycatch mortality continues. While many of the threats to Gulf sturgeon have been ameliorated or reduced due to the existing regulatory mechanisms, such as the moratorium on directed fisheries, bycatch is not currently being addressed. Therefore, the loss of Gulf sturgeon as bycatch likely continues.

3.2.9 Giant Manta Ray

We listed the giant manta ray (*Manta birostris*) as threatened under the ESA (83 FR 2916, January 22, 2018) and determined that the designation of critical habitat is not prudent on (84 FR 66652, December 5, 2019). On December 4, 2019, we published a recovery outline for the giant manta ray (NMFS 2019b), which 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 2 structures called cephalic lobes that extend and help to introduce water into the mouth for feeding activities (making them the only vertebrate animals with 3 paired appendages). Giant manta rays have 2 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 2 mirror image right-angle triangles, creating a T-shape on the upper shoulders.

The giant manta ray is found worldwide in tropical and subtropical oceans and in productive coastal areas. 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 (CITES 2013; Gudger 1922; Kashiwagi et al. 2010; Moore 2012). In the Southern Hemisphere, the species occurs as far south as Peru, Uruguay, South Africa, New Zealand and French Polynesia (CITES 2013; Mourier 2012). Within this 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 7) (Kashiwagi et al. 2011; Marshall et al. 2009), as may occasionally occur within estuaries (e.g., lagoons and bays).

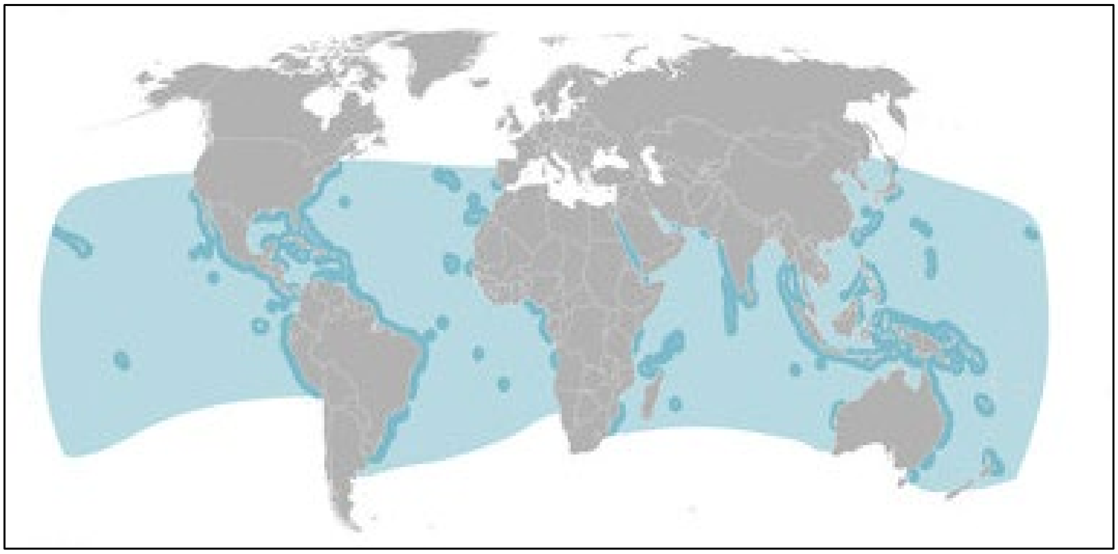

Figure 7. The Extent of Occurrence (dark blue) and Area of Occupancy (light blue) based on species distribution (Lawson et al. 2017).

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; Girondot et al. 2015; Graham et al. 2012; Stewart et al. 2016). 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 inlets, with use of these waters as potential nursery grounds (J. Pate, Florida Manta Project, unpublished data; Adams and Amesbury 1998; Medeiros et al. 2015; Milessi and Oddone 2003).

Giant manta rays are known to aggregate in various locations around the world in groups usually ranging from 100-1,000 (Graham et al. 2012; Notarbartolo di Sciara and Hillyer 1989; Venables 2013). These aggregation locations function as feeding sites, cleaning stations, or sites where courtship interactions take place (Graham et al. 2012; Heinrichs et al. 2011; 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 (Heinrichs et al. 2011; Notarbartolo di Sciara and Hillyer 1989). 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 Hoona Bays in Hawaii, appear to exhibit learned behavior. These manta rays learned to associate artificially 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 and the surrounding region might represent the first documented nursery habitat for giant manta ray (Stewart et al. 2018). Stewart et al. (2018) found the Flower Garden Banks National Marine Sanctuary 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 Flower Garden Banks National Marine Sanctuary 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 southeast 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 200-450 m depths (Rubin et al. 2008; Stewart et al. 2016) and are capable of diving to depths exceeding 1,000 m (A. Marshall et al., unpublished 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 (Burgess et al. 2016; Couturier et al. 2013).

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). Based on field observations it was previously assumed that giant manta rays feed predominantly during the day on surface zooplankton, however, results from recent studies (Burgess et al. 2016; Couturier et al. 2013) 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 2 to 3 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 10 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 (Marshall and Holmberg 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 U.S. Atlantic and Caribbean, giant manta ray sightings are concentrated along the east coast as far north as New Jersey, within the Gulf of Mexico, and off the coasts of the U.S. Virgin Islands and Puerto Rico. Because most sightings of the species have been opportunistic during other surveys, researchers are still unsure what attracts giant manta rays to certain areas and not others and where they go for the remainder of the time (84 FR 66652, December 5, 2019).

The available sightings data indicate that giant manta rays occur regularly along Florida's east coast. In 2010, Georgia Aquarium began conducting aerial surveys for giant manta rays. The surveys are conducted in spring and summer and run from the beach parallel to the shoreline (0-2.5 nm), from St. Augustine Beach Pier to Flagler Beach Pier, Florida. The numbers, location, and peak timing of the manta rays to this area varies by year (H. Webb unpublished data). In addition, off southeast Florida, juvenile giant manta rays have also been regularly observed in inshore waters. 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 nm) (J. Pate, MMF, pers. comm. to M. Miller, NMFS OPR, 2018). Results from these surveys indicate that juvenile manta rays are present in these waters for the majority of the year (observations span from May to December), with re-sightings data that suggest some manta rays may 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 (Convention on Migratory Species 2014; O'Malley et al. 2013). 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 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 (Chassot et al. 2008; Romanov 2002). 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 longline, pelagic bottom longline, and trawl fisheries. Incidental capture of giant manta ray is also a rare occurrence in the elasmobranch catch within U.S. Atlantic and Gulf of Mexico, with the majority that are caught released alive. In addition to directed harvest and bycatch in commercial fisheries, the giant manta ray is incidentally captured by recreational fishers using vertical line (i.e., handline, bandit gear, and rod-and-reel). Researchers frequently report giant manta rays having evidence of recreational gear interactions along the east coast of Florida (e.g., manta rays with embedded fishing hooks and trailing monofilament line) (J. Pate, Florida Manta Project, unpublished data). Internet searches also document 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). In addition, 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 is expected to be low, given that we have no reports of recreational fishers retaining giant manta ray. However, bycatch in recreational fisheries remains a potential threat to the species.

Vessel strikes can injure or kill giant manta rays, decreasing fitness or contributing to non-natural mortality (Couturier et al. 2012; Deakos et al. 2011). Giant manta rays do not surface to breath, 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). They show little fear toward vessels, which can also make them extremely vulnerable to vessel strikes (Deakos 2010; C. Horn. NMFS, personal observation). Five giant manta rays were reported to have been struck by vessels from 2016 through 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 (i.e., via strandings) have been documented. This lack of documented mortalities could also be the result of other factors that influence carcass detection (e.g., 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).

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. 2019). 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. 2019). 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 (Marshall and Holmberg 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. While the ingestion of plastics is likely to negatively affect the health of the species, the levels of microplastics in manta ray feeding grounds and frequency of ingestion are presently being studied to evaluate the impact on these species (Germanov et al. 2019).

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 2 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 2010). 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 (Braun et al. 2015; Convention on Migratory Species 2014; Couturier et al. 2012; Deakos et al. 2011; Germanov and Marshall 2014; Heinrichs et al. 2011). 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.

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 their 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 Klimovich 2017).

3.2.10 Smalltooth Sawfish

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

Species Description and Distribution
The smalltooth sawfish is a tropical marine and estuarine elasmobranch. It is a batoid with a long, narrow, flattened, rostral blade (rostrum) lined with a series of transverse teeth along either edge. In general, smalltooth sawfish inhabit shallow coastal waters of the Atlantic Ocean (Dulvy et al. 2016) and feed on a variety of fish (e.g., mullet, jacks, and ladyfish) (Simpfendorfer 2001; Poulakis et al. 2017).

Although this species is reported throughout the tropical Atlantic, we identified smalltooth sawfish from the Southeast United States as a DPS due to the physical isolation of this population from others, the differences in international management of the species, and the significance of the U.S. population in relation to the global range of the species (see 68 FR 15674). Within the U.S., smalltooth sawfish have historically been captured in estuarine and coastal waters from North Carolina southward through Texas, although peninsular Florida has been the region of the United States with the largest number of recorded captures (NMFS 2018). Recent records indicate there is a resident reproducing population of smalltooth sawfish in south and southwest Florida from Charlotte Harbor through the Florida Keys, which is also the last U.S. stronghold for the species (Poulakis and Seitz 2004; Seitz and Poulakis 2002; Simpfendorfer and Wiley 2005). Water temperatures (no lower than 8-12°C) and the availability of appropriate coastal habitat (shallow, euryhaline waters and red mangroves) are the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic. Most specimens captured along the Atlantic coast north of Florida are large juveniles or adults (over 10 ft) that likely represent seasonal migrants, wanderers, or colonizers from a historical Florida core population to the south, rather than being members of a continuous, even-density population (Bigelow and Schroeder 1953b).

Life History Information
Smalltooth sawfish mate in the spring and early summer (Grubbs unpublished data; Poulakis unpublished data). Fertilization is internal and females give birth to live young. Evidence suggests a gestation period of approximately 12 months (Feldheim et al. 2017, Gelsleichter unpublished data) and females produce litters of 7-14 young (Gelsleichter unpublished data;

Feldheim et al. 2017). Females have a biennial reproductive cycle (Feldheim et al. 2017) and parturition (act of giving birth) occurs nearly year round though peaking in spring and early summer (March-July) (Poulakis et al. 2011, Carlson unpublished data). Smalltooth sawfish are approximately 26-31 in (64-80 cm) at birth (Poulakis et al. 2011; Bethea et al. 2012) and may grow to a maximum length of approximately 16 ft (5 m) (Grubbs unpublished data, Brame et al. 2019). Simpfendorfer et al. (2008) report rapid juvenile growth for smalltooth sawfish for the first 2 years after birth, with stretched TL increasing by an average of 25-33 in (65-85 cm) in the first year and an average of 19-27 in (48-68 cm) in the second year. Uncertainty remains in estimating post-juvenile growth rates and age at maturity, yet, recent advances indicate maturity at 7-11 years (Carlson and Simpfendorfer 2015) at lengths of approximately 3.4 m for males and 3.5-3.7 m for females (Gelsleichter unpublished data).

There are distinct differences in habitat use based on life history stage as the species shifts use through ontogeny. Juvenile smalltooth sawfish less than 2.2 m in length inhabit the shallow euryhaline waters (i.e., variable salinity) of estuaries, and can be found in sheltered bays, dredged canals, along banks and sandbars, and in rivers (NMFS 2000). These juveniles are often closely associated with muddy or sandy substrates, and shorelines containing red mangroves, *Rhizophora* mangle (Simpfendorfer 2001; Simpfendorfer 2003; Simpfendorfer et al. 2010; Poulakis et al. 2011; Poulakis et al. 2013; Hollensead et al. 2016; Hollensead et al. 2018). Simpfendorfer et al. (2010) indicated the smallest juveniles (YOY juveniles measuring < 1 m in length) generally used the shallowest water (depths less than 0.5 m [1.64 ft]), had small home ranges (4,264-4,557 m^2), and exhibited high levels of site fidelity. Although small juveniles exhibit high levels of site fidelity for specific nursery habitats for periods of time lasting up to 3 months (Wiley and Simpfendorfer 2007), they do undergo small movements coinciding with changing tidal stages. These movements often involve moving from shallow sandbars at low tide to within red mangrove prop roots at higher tides (Simpfendorfer et al. 2010)—behavior likely to reduce the risk of predation (Simpfendorfer 2006). As juveniles increase in size, they begin to expand their home ranges (Simpfendorfer et al. 2010; Simpfendorfer et al. 2011), eventually moving to more offshore habitats where they likely feed on larger prey as they continue to mature.

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

The juvenile "hotspots" may be of further significance following the findings of female philopatry (Feldheim et al. 2017). More specifically, Feldheim et al. (2017) found that female sawfish return to the same parturition (birthing) sites over multiple years (parturition site fidelity). We expect that these parturition sites align closely with the juvenile "hotspots" given the high fidelity shown by the smallest size/age classes of sawfish to specific nursery areas. Therefore, disturbance of these nursery areas could have wide-ranging effects on the sawfish population if it were to disrupt future parturition.

While adult smalltooth sawfish may also use the estuarine habitats used by juveniles, they are commonly observed in deeper waters along the coasts. Poulakis and Seitz (2004) noted that nearly half of the encounters with adult-sized smalltooth sawfish in Florida Bay and the Florida Keys occurred in depths from 200-400 ft (70-122 m) of water. Similarly, Simpfendorfer and Wiley (2005) reported encounters in deeper waters off the Florida Keys, and observations from both commercial longline fishing vessels and fishery-independent sampling in the Florida Straits report large smalltooth sawfish in depths up to 130 ft (~40 m) (ISED 2014). Yet, current field studies show adult smalltooth sawfish also use shallow estuarine habitats within Florida Bay and the Everglades (Grubbs unpublished data). Further, we expect that females return to shallow estuaries during parturition (when adult females return to shallow estuaries to give birth).

Status and Population Dynamics

Based on the contraction of the species' geographic range, we expect that the population to be a fraction of its historical size. Few long-term abundance data exist for the smalltooth sawfish, however, making it very difficult to estimate the current population size. Despite the lack of scientific data, recent encounters with YOY, older juveniles, and sexually mature smalltooth sawfish indicate that the U.S. population is currently reproducing (Seitz and Poulakis 2002; Simpfendorfer 2003, Grubbs unpublished data, Feldheim et al. 2017). The abundance of juveniles publically encountered by anglers and boaters, including very small individuals, suggests that the population remains viable (Simpfendorfer and Wiley 2004). Further, data analyzed from Everglades National Park (ENP) as part of an established fisheries-dependent monitoring program (angler interviews) indicated a slightly increasing trend in juvenile abundance within the park over the past decade (Carlson and Osborne 2012; Carlson et al. 2007). Similarly, preliminary results of juvenile smalltooth sawfish sampling programs in both ENP and Charlotte Harbor indicate the juvenile population is at least stable and possibly increasing (Poulakis unpublished data, Carlson unpublished data).

Using a demographic approach and life history data for smalltooth sawfish and similar species from the literature, Simpfendorfer (2000) estimated intrinsic rates of natural population increase for the species at 0.08-0.13 per year and population doubling times from 5.4-8.5 years. These low intrinsic rates[11] of population increase, suggest that the species is particularly vulnerable to excessive mortality and rapid population declines, after which recovery may take decades. Carlson and Simpfendorfer (2015) constructed an age-structured Leslie matrix model for the

[11] The rate at which a population increases in size if there are no density-dependent forces regulating the population.

U.S. population of smalltooth sawfish, using updated life history information, to determine the species' ability to recover under scenarios of variable life history inputs and the effects of bycatch mortality and catastrophes. As expected, population growth was highest (λ=1.237 yr-1) when age-at-maturity was 7 yr and decreased to 1.150 yr-1 when age-at-maturity was 11 yr. Despite a high level of variability throughout the model runs, in the absence of fishing mortality or catastrophic climate effects, the population grew at a relatively rapid rate approaching carrying capacity in 40 years when the initial population was set at 2,250 females or 50 years with an initial population of 600 females. Carlson and Simpfendorfer (2015) concluded that smalltooth sawfish in U.S. waters appear to have the ability to recover within the foreseeable future based on a model relying upon optimistic estimates of population size, lower age-at-maturity and the lower level of fisheries-related mortality. Another analysis was less optimistic based on lower estimates of breeding females in the Caloosahatchee River nursery (Chapman unpublished data). Assuming similar numbers of females among the 5 known nurseries, that study would suggest an initial breeding population of only 140-390 females, essentially half of the initial population considered by Carlson and Simpfendorfer (2015). A smaller initial breeding population would extend the time to reach carrying capacity.

Threats

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

<u>Bycatch Mortality</u>

Bycatch mortality is cited as the primary cause for the decline in smalltooth sawfish in the United States (NMFS 2010). While there has never been a large-scale directed fishery, smalltooth sawfish become easily entangled in fishing gear (gill nets, otter trawls, trammel nets, and seines) directed at other commercial species, often resulting in serious injury or death (NMFS 2009b). This has been historically reported in Florida (Snelson and Williams 1981), Louisiana (Simpfendorfer 2002), and Texas (Baughman 1943). For instance, one fisher interviewed by Evermann and Bean (1897) reported taking an estimated 300 smalltooth sawfish in just one netting season in the Indian River Lagoon, Florida. In another example, smalltooth sawfish landings data gathered by Louisiana shrimp trawlers from 1945-1978, which contained both landings data and crude information on effort (number of vessels, vessel tonnage, number of gear units), indicated declines in smalltooth sawfish landings from a high of 34,900 lbs in 1949 to less than 1,500 lbs in most years after 1967. The Florida net ban passed in 1995 has led to a reduction in the number of smalltooth sawfish incidentally captured, "…by prohibiting the use of gill and other entangling nets in all Florida waters, and prohibiting the use of other nets larger than 500 square ft in mesh area in nearshore and inshore Florida waters" [12] (FLA. CONST. art. X, § 16). However, the threat of bycatch currently remains in commercial fisheries (e.g., Gulf of Mexico and South Atlantic shrimp fisheries, federal shark fisheries of the South Atlantic, and the Gulf of Mexico reef fish fishery), though anecdotal information collected by our port agents suggest smalltooth sawfish captures are now rare.

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

<u>Habitat Loss</u>

Modification and loss of smalltooth sawfish habitat, especially nursery habitat, is another contributing factor in the decline of the species. Activities such as agricultural and urban development, commercial activities, dredge-and-fill operations, boating, erosion, and diversions of freshwater runoff contribute to these losses (SAFMC 1998). Large areas of coastal habitat were modified or lost between the mid-1970s and mid-1980s within the U.S. (Dahl and Johnson 1991). Since then, rates of loss have decreased, but habitat loss continues. From 1998-2004, approximately 64,560 acres of coastal wetlands were lost along the Atlantic and Gulf coasts of the United States, of which approximately 2,450 acres were intertidal wetlands consisting of mangroves or other estuarine shrubs (Stedman and Dahl 2008). Further, Orlando et al. (1994) analyzed 18 major southeastern estuaries and recorded over 703 mi of navigation channels and 9,844 mi of shoreline with modifications. In Florida, coastal development often involves the removal of mangroves and the armoring of shorelines through seawall construction. Changes to the natural freshwater flows into estuarine and marine waters through construction of canals and other water control devices have had other impacts: altered the temperature, salinity, and nutrient regimes; reduced both wetlands and submerged aquatic vegetation; and degraded vast areas of coastal habitat utilized by smalltooth sawfish (Gilmore 1995; Reddering 1988; Whitfield and Bruton 1989). While these modifications of habitat are not the primary reason for the decline of smalltooth sawfish abundance, it is likely a contributing factor and almost certainly hampers the recovery of the species. Juvenile sawfish and their nursery habitats are particularly likely to be affected by these kinds of habitat losses or alternations, due to their affinity for shallow, estuarine systems. Prohaska et al. (2018) showed that juvenile smalltooth sawfish within the anthropogenically altered Charlotte Harbor estuary have higher metabolic stress compared to those collected from more pristine nurseries in the Everglades. Although many forms of habitat modification are currently regulated, some permitted direct and/or indirect damage to habitat

[12] "nearshore and inshore Florida waters" means all Florida waters inside a line 3 mi seaward of the coastline along the Gulf of Mexico and inside a line 1 mi seaward of the coastline along the Atlantic Ocean.

from increased urbanization still occurs and is expected to continue to threaten survival and recovery of the species in the future.

Life History Limitations

The smalltooth sawfish is also limited by its life history characteristics as a relatively slow-growing, late-maturing, and long-lived species. Animals with this life history strategy are usually successful in maintaining small, persistent population sizes in constant environments, but are particularly vulnerable to increases in mortality or rapid environmental change (NMFS 2000). The combined characteristics of this life history strategy result in a very low intrinsic rate of population increase (Musick 1999) that make it slow to recover from any significant population decline (Simpfendorfer 2000).

Stochastic Events

Although stochastic events such as aperiodic extreme weather and HABs are expected to affect smalltooth, we are currently uncertain of their impact. A strong and prolonged cold weather event in January 2010 resulted in the mortality of at least 15 juvenile and 1 adult sawfish (Poulakis et al. 2011; Scharer et al. 2012), and led to far fewer catches in directed research throughout the remainder of the year (Bethea et al. 2011). Another less severe cold front in 2011 did not result in any known mortality but did alter the typical habitat use patterns of juvenile sawfish within the Caloosahatchee River. Since surveys began, 2 hurricanes have made direct landfall within the core range of U.S. sawfish. While these storms denuded mangroves along the shoreline and created hypoxic water conditions, we are unaware of any direct effects to sawfish. Just prior to the passage of the most recent hurricane (Hurricane Irma in 2017), acoustically tagged sawfish moved away from their normal shallow nurseries and then returned within a few days (Poulakis unpublished data; Carlson unpublished data). HABs have occurred within the core range of smalltooth sawfish and affected a variety of fauna including sea turtles, fish, and marine mammals, but to date no sawfish mortalities have been reported.

Current Threats

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

In addition to the anthropogenic effects mentioned previously, changes to the global climate are likely to be a threat to smalltooth sawfish and the habitats they use. The IPCC has stated that global climate change is unequivocal and its impacts to coastal resources may be significant (IPCC 2007; IPCC 2013). Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, changes in the amount and timing of precipitation, and changes in air and water temperatures (EPA 2012; NOAA 2012). The impacts to smalltooth sawfish cannot, for the most part, currently be predicted with any degree of certainty, but we can project some effects to the coastal habitats where they reside. Red mangroves and shallow, euryhaline waters will be directly impacted by climate change through sea level rise, which is expected to increase 0.45 to 0.75 m by 2100 (IPCC 2013). Sea level rise will impact mangrove resources, as sediment surface elevations for mangroves will not keep pace with conservative projected rates of elevation in sea level (Gilman et al. 2008). Sea level increases will also affect the amount of shallow water available for juvenile smalltooth sawfish nursery habitat, especially in areas where there is shoreline armoring (e.g., seawalls). Further, the changes in precipitation coupled with sea level rise may also alter salinities of coastal habitats, reducing the amount of available smalltooth sawfish nursery habitat.

4 ENVIRONMENTAL BASELINE

By regulation, environmental baselines for Opinions include the past and present impacts of all state, federal, or private actions and other human activities in the action area. We identify the anticipated impacts of all proposed federal projects in the specific action area of the consultation at issue, that have already undergone formal or early Section 7 consultation as well as the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). Focusing on the impacts of the activities in the action area specifically, allows us to assess the prior experience and condition of threatened and endangered species, and areas of designated critical habitat that occur in an action area, and that will be exposed to effects from the action under consultation. This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and the ecosystem, within the action area. The environmental baseline is a “snapshot” of a species’ health at a specified point in time. It does not include the effects of the action under review in this consultation. The environmental baseline for this Opinion includes the effects of several activities that may affect the survival and recovery of loggerhead, green, Kemp’s ridley, leatherback, and hawksbill sea turtles, as well as Atlantic and Gulf sturgeon, smalltooth sawfish, and giant manta ray within the action area.

4.1 Status of Species within the Action Area

The status of the listed species in the action area, as well as the threats to each of these species, is supported by the species accounts in Section 3. As stated in Section 2.2, the proposed action would occur throughout the Gulf and South Atlantic EEZ, and adjacent marine and tidal state waters of the Gulf and South Atlantic area (i.e., from the Mexico-Texas border to the North Carolina-Virginia border).

4.2.1 Factors Affecting Listed Species within the Action Area

4.2.1 Federal Actions

We have undertaken a number of Section 7 consultations to address the effects of federally-permitted fisheries and other federal actions on threatened and endangered species, and when appropriate, has authorized the incidental taking of these species. Each of those consultations sought to minimize the adverse effects of the action on these affected species. The summary below of federal actions and the effects these actions have had on ESA-listed species includes only those federal actions in the action area, which have already concluded or are currently undergoing formal Section 7 consultation.

4.2.1.1 Fisheries

Threatened and endangered sea turtle and fish species are adversely affected by fishing gears used throughout the action area. Trawl gear has been documented to interact with sea turtles, as well as sturgeon, smalltooth sawfish, and giant manta ray. The southeast U.S. shrimp fishery is being analyzed in this document as part of the proposed action so only the fisheries' past effects are considered as part of this environmental baseline. Sea turtles are also affected by gillnet, pelagic and bottom longline, other types of hook-and-line gear, and pot fisheries. Sturgeon have been entangled in gillnet gear, with the greatest number of captures and highest mortality rates occurring in sink gillnets. Smalltooth sawfish are adversely affected by hook-and-line and gillnet gear. For all federal fisheries for which there is a FMP, impacts have been evaluated through Section 7 consultation. Some of these consultations resulted in subsequent rulemaking to reduce the impacts of the specific fisheries on sea turtle populations. Examples include additional monitoring of and TED requirements in the southeast U.S. shrimp fisheries, as well as gear limitations and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality in Atlantic highly migratory species (HMS) fisheries and reef fish fisheries. All Opinions had an ITS and determined that fishing activities, as considered (i.e., with conservation requirements) would not jeopardize any listed species. Current anticipated take levels associated with these fisheries are presented in Appendix 1; the take levels reflect the impact on listed species of each activity anticipated from the date of the ITS forward in time. A summary of each of consultation is provided below; more detailed information can be found in the respective fisheries' most recent Opinions, which are also cited in the corresponding sections below, and are incorporated herein by reference.

Atlantic Pelagic Longline (PLL) Fisheries
Atlantic PLL fisheries targeting swordfish and tuna are also known to incidentally capture and kill large numbers of loggerhead (pelagic juvenile loggerhead sea turtles) and leatherback sea turtles. U.S. PLL fishers began targeting HMS in the Atlantic Ocean in the early 1960s. The fisheries are comprised of 5 relatively distinct segments, including: the Gulf yellowfin tuna fishery; southern Atlantic (Florida East Coast to Cape Hatteras) swordfish fishery; Mid-Atlantic and New England swordfish and bigeye tuna fishery; U.S. Atlantic Distant Water swordfish fishery; and the Caribbean tuna and swordfish fishery. Although 2 fishery segments occur in the action area, fishing occurs farther offshore than where shrimp trawling occurs.

Over the past 2 decades, we have conducted numerous consultations on these fisheries, some of which required RPAs to avoid jeopardy of loggerhead and/or leatherback sea turtles. The estimated historical total number of loggerhead and leatherback sea turtles caught between 1992-2002 (all geographic areas) is 10,034 loggerhead and 9,302 leatherback sea turtles of which 81 and 121 were estimated to be dead when brought to the vessel (NMFS 2004). This does not account for post-release mortalities, which historically were likely substantial.

We reinitiated consultation in 2003 on PLL fisheries as a result of exceeded incidental take levels for loggerheads and leatherbacks (NMFS 2004). The resulting June 1, 2004, Opinion stated the long-term operation of this segment of the fisheries was likely to jeopardize the continued existence of leatherback sea turtles, but RPAs were implemented allowing for the authorization of PLL fishing that would not jeopardize leatherback sea turtles.

On July 6, 2004, we published a final rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic PLL fishery (69 FR 40734). The management measures include mandatory circle hook and bait requirements, and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality. The rulemaking, based on the results of the 3-year Northeast Distant Closed Area research experiment and other available sea turtle bycatch reduction studies, is expected to have significantly benefitted endangered and threatened sea turtles by reducing mortality attributed to this fishery.

On March 31, 2014, the HMS Management Division requested that we reinitiate formal Section 7 consultation for the Atlantic PLL fishery based on the availability of information revealing effects of the action that may affect listed species in a manner or to an extent not previously considered (see 50 CFR 402.16 (b)). Specifically, the request is based on information indicating that the net mortality rate and total mortality estimates for leatherback sea turtles specified in the reasonable and prudent alternative were exceeded (although the take level specified in the ITS has not been exceeded), changes in information about leatherback and loggerhead sea turtle populations, and new information about sea turtle mortality associated with PLL gear.

On May 15, 2020, we completed an Opinion on the PLL fishery as managed under the 2006 Consolidated Atlantic HMS FMP to revisit the effects of the fishery on leatherback and the NWA DPS of loggerhead sea turtles (including take estimates for both species), and to address potential effects on the newly listed Central and Southwest Atlantic DPSs of scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray; effects on sperm whale, NA DPS of green sea turtle, and olive ridley sea turtle were also evaluated. The Opinion concluded the entire proposed action was not likely to jeopardize the continued existence of any listed species, and an ITS was provided.

Atlantic HMS Fisheries (Other than PLL)
Atlantic HMS commercial directed shark fisheries also adversely affect sea turtles via capture and/or entanglement in the action area. The commercial component uses bottom longline and gillnet gear. Bottom longline is the primary gear used to target large coastal sharks (LCS) in the Gulf of Mexico. The largest concentration of bottom longline fishing vessels is found along the central Gulf coast of Florida, with the John's Pass/Madeira Beach area considered the center of directed shark fishing activities. Gillnets are the dominant gear for catching small coastal sharks; most shark gillnetting occurs off southeast Florida. Growing demand for shark and shark

products encouraged expansion of the commercial shark fishery through the 1970s and 1980s. As catches accelerated through the 1980s, shark stocks started to show signs of decline. Peak commercial landings of large coastal and pelagic sharks were reported in 1989.

We have managed Atlantic LCS, small coastal sharks, and pelagic sharks since 1993 under an FMP for Atlantic sharks. Observation of directed HMS shark fisheries has been ongoing since 1994, but a mandatory program was not implemented until 2002. Sea turtle bycatch in the fishery has primarily been neritic juvenile and adult loggerhead sea turtles, but leatherback sea turtles captures have also been observed, as well as a few observations of unidentified species of turtles. Between 1994 and 2002, the program covered 1.6% of all hooks, and over that time period caught 31 loggerhead sea turtles, 4 leatherback sea turtles, and 8 unidentified with estimated annual average take levels of 30, 222, and 56, respectively.

In 2008, we completed a Section 7 consultation on the authorization of directed Atlantic HMS shark fisheries under the Consolidated HMS FMP, including Amendment 2 (NMFS 2008). To protect declining shark stocks, Amendment 2 sought to greatly reduce the fishing effort in the commercial component of the fishery. These effort reductions are believed to have greatly reduced the interactions between the commercial component of the fishery and sea turtles. Amendment 2 to the Consolidated HMS FMP (73 FR 35778, June 24, 2008, corrected at 73 FR 40658, July 15, 2008) established, among other things, a shark research fishery to maintain time series data for stock assessments and to meet our 2009 research objectives. The shark research fishery permits authorize participation in the shark research fishery and the collection of sandbar and non-sandbar LCS from federal waters in the Atlantic Ocean, Gulf of Mexico, and Caribbean Sea for the purposes of scientific data collection subject to 100% observer coverage. Commercial vessels not participating in the shark research fishery are subject to 4-6% observer coverage and may only land non-sandbar LCS, SCS, and pelagic sharks subject to the retention limits and quotas per 50 CFR 635.24 and 635.27, respectively.

During 2007-2011, 10 sea turtle captures (all loggerheads) were observed on bottom longline gear in the sandbar shark research fishery and 5 were captured outside the research fishery. The 5 non-research fishery captures were extrapolated to the entire fishery, providing a bycatch estimate of 45.6 sea turtles (all loggerheads) for non-sandbar shark research fishery from 2007-2010 (Carlson and Richards 2011). No sea turtle captures were observed in the non-research fishery in 2011 (NMFS unpublished data). Sixteen smalltooth sawfish captures were observed in the sandbar shark research fishery from 2007-2011, and 6 were captured outside the research fishery (Carlson and Richards 2011; NMFS unpublished data); one capture in the shark bottom longline fishery resulted in mortality. The 6 non-research fishery captures were extrapolated to the entire fishery, providing an estimate of 17.3 total smalltooth sawfish captures for non-sandbar shark research fishery. Since the research fishery has a 100% observer coverage requirement, observed interactions were not extrapolated (Carlson and Richards 2011).

On December 12, 2012, we completed a consultation on the operation of shark fisheries and Amendments 3 and 4 to the Consolidated HMS FMP (NMFS 2012b). The 2012 Opinion analyzed the potential adverse effects from the smoothhound shark fishery on sea turtles for the first time. Few smoothhound shark trips have been observed and no sea turtle or smalltooth sawfish captures have been documented in the smoothhound shark fishery. The Opinion concluded the entire proposed action was not likely to jeopardize the continued existence of sea turtles, Atlantic sturgeon, or smalltooth sawfish, and an ITS was provided.

On January 10, 2020, we completed an Opinion on the operation of Atlantic HMS fisheries (excluding the PLL fishery) as carried out under the 2006 Consolidated Atlantic HMS FMP, as amended. Consultation was initially reinitiated to address potential effects on the newly listed Central and Southwest Atlantic DPSs of scalloped hammerhead shark, and 7 species of corals. Additionally the subsequent designation of critical habitat for the NWA DPS of loggerhead sea turtle, as well as the subsequent listing of 2 DPSs of green sea turtles as threatened, Nassau grouper as threatened, Bryde's whale as endangered, oceanic whitetip shark as threatened, and giant manta ray as threatened were ultimately added to the scope of this consultation.

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 bycatch is lower than in the PLL fishery. The January 10, 2020 Opinion concluded the entire proposed action was not likely to jeopardize the continued existence of any listed species, and an ITS was provided.

On May 15, 2020, we reissued the aforementioned January 10, 2020, Opinion to include discretionary conservation recommendations that were mistakenly omitted, and to correct a few minor non-substantive errors (NMFS 2020b). The May 15, 2020, Opinion supersedes the January 10, 2020 Opinion.

Gulf of Mexico Reef Fish Fishery
The Gulf of Mexico reef fish fishery uses 2 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). Trap gear was phased-out completely by February 2007, but prior to that the gear likely resulted in a few sea turtle and smalltooth sawfish entanglements.

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, with approximately 107 captures and 41 mortalities annually, all species combined, for the entire fishery (NMFS 2005b). The hook-and-line components of the fishery have likely always had the most adverse effects on smalltooth sawfish. In 2008, our SEFSC observer program and subsequent analyses indicated that the overall amount and extent of incidental take for sea turtles specified in the ITS of the 2005 Opinion on the reef fish fishery had been severely exceeded by the bottom longline component (approximately 974 captures and at least 325 mortalities estimated for the period July 2006-2007).

In response, we 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 6 months pending the implementation of a long-term management strategy. The 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, we 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 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, we reinitiated consultation to address the DWH 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 ITS. 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 species, and an ITS was provided (NMFS 2011b).

South Atlantic Snapper-Grouper Fishery
The South Atlantic snapper-grouper fishery uses spear and powerheads, black sea bass pots, 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 (i.e., handline, bandit gear, and rod-and-reel). The most recent consultation on the fishery was completed in 2016 (NMFS 2016), which concluded the proposed action was not likely to jeopardize the continued existence of the North Atlantic right whale, NWA DPS of the loggerhead sea turtle, leatherback sea turtle, Kemp's ridley sea turtle, NA or SA DPS of the green sea turtle, hawksbill sea turtle, U.S. DPS of the smalltooth sawfish, and Nassau grouper, and an ITS was provided.

Coastal Migratory Pelagics (CMP) Fishery
We completed a Section 7 consultation on the authorization of CMP fishery in the Gulf of Mexico and South Atlantic (NMFS 2007b). Commercial fishers target king and Spanish mackerel with hook-and-line (i.e., handline, rod-and-reel, and bandit), gillnet, and cast net gears. Recreational fishers use only rod-and-reel gear. Trolling is the most common hook-and-line fishing technique used by both commercial and recreational fishers. A winter troll fishery

operates along the east and south Gulf coast. Although run-around gillnets accounted for the majority of the king mackerel catch from the late 1950s through 1982, handline gear has been the predominant gear used in the commercial king mackerel fishery since 1993 (NMFS 2007b). The gillnet fishery for Gulf king mackerel is restricted to the use of "run-around" gillnets in Monroe and Collier Counties in January. Run-around gillnets are still the primary gear used to harvest Spanish mackerel, but the fishery is relatively small because Spanish mackerel are typically more concentrated in state waters where gillnet gear is prohibited. The 2007 Opinion concluded that green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles, as well as smalltooth sawfish may be adversely affected by the gillnet component of the fishery. The authorization of the fishery was not expected to jeopardize the continued existence of any of these species, and an ITS was provided.

A June 18, 2015 Opinion, amended on November 18, 2017 via a 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 listed sea turtle species, and an ITS was provided.

Spiny Lobster Fishery

We completed a Section 7 consultation on the Gulf and South Atlantic Spiny Lobster FMP on August 27, 2009 (NMFS 2009d). The commercial component of the fishery consists of diving, bully net and trapping sectors; recreational fishers are authorized to use bully net and hand-harvest gears. Of the gears used, only traps are expected to result in adverse effects on sea turtles and smalltooth sawfish. The consultation determined the authorization of the fishery would not jeopardize any listed species. An ITS was issued for takes in the commercial trap sector of the fishery. Fishing activity using traps is limited to waters off south Florida and, although the FMP does authorize the use of traps in federal waters, historic and current effort is very limited. Thus, potential adverse effects on sea turtles are believed to also be very limited (e.g., no more than a couple sea turtle or smalltooth sawfish entanglements occurring annually).

Stone Crab Fishery

We completed a Section 7 consultation on the Gulf of Mexico Stone Crab FMP on September 28, 2009 (NMFS 2009e). The commercial component of the fishery is traps; recreational fishers use traps or dive (i.e., hand harvest) for stone crabs. Of the gears used, only commercial traps are expected to result in adverse effects on sea turtles or smalltooth sawfish. The number of commercial traps actually in the water is very difficult to estimate, and the number of traps used recreationally is unquantifiable with any degree of accuracy. The consultation determined the authorization of the fishery was likely to adversely affect sea turtles and smalltooth sawfish, but would not jeopardize their continued existence; an ITS was issued for takes in the commercial trap sector of the fishery. On October 28, 2011, we repealed the federal FMP for this fishery, and the fishery is now managed exclusively by the state of Florida. Since the State of Florida has essentially been the lead management agency for the state and federal fishery for some time, little change in how the fishery operates or amount of the effort occurring in the fishery is expected because of the repeal of the federal FMP. Therefore, the anticipated adverse effects described in the Opinion completed before the repeal of the federal FMP are expected to continue to occur to listed species.

Dolphin/Wahoo Fishery
The South Atlantic FMP for the dolphin/wahoo fishery was approved in December 2003. We conducted a formal Section 7 consultation to consider the effects on sea turtles of authorizing fishing under the FMP (NMFS 2003). 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.

4.2.1.2 Federal Dredging Activity

Marine dredging vessels are common within U.S. coastal waters, and construction and maintenance of federal navigation channels and dredging in sand mining sites (borrow areas) have been identified as sources of sea turtle and sturgeon mortality. Hopper dredges 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. Likewise, mechanical dredges have also been documented to kill Atlantic and Gulf sturgeon (Dickerson 2005). Dickerson (2013) summarized observed takings of 26 sturgeon from dredging activities conducted by the USACE observed between October 1990 and January 2013 (3 Gulf and 23 Atlantic). Of the 3 types of dredges included (hopper, clam, and pipeline) in the report, hopper dredges captured the most sturgeon.

To reduce take of listed species, relocation trawling may be utilized to capture and move sea turtles and sturgeon. In relocation trawling, a boat equipped with nets precedes the dredge to capture sturgeon and then releases the animals out of the dredge pathway, thus avoiding lethal take. Seasonal in-water work periods, when the species is absent from the project area, also assists in reducing incidental take.

Dredging activities can also pose significant impacts to aquatic ecosystems utilized by sturgeon, including: 1) direct removal/burial of organisms; 2) turbidity/siltation effects; 3) contaminant re-suspension; 4) noise/disturbance; 5) alterations to hydrodynamic regime and physical habitat; and 6) loss of riparian habitat (Chytalo 1996; Winger et al. 2000). Dredging operations may destroy benthic feeding areas, disrupt spawning migrations, and re-suspend fine sediments causing siltation over required substrate in spawning habitat. Because sturgeon are benthic omnivores, the modification of the benthos affects the quality, quantity, and availability of prey.

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 or sturgeon.

In summary, dredging and disposal to maintain navigation channels, and removal of sediments for beach renourishment occurs frequently and throughout the range of sea turtles, sturgeon, and within Gulf sturgeon critical habitat annually. This activity has, and continues to, threaten the species and affect its designated critical habitat.

We originally completed regional Opinions on the impacts of USACE's hopper-dredging operation in 1997 for dredging along the South Atlantic (i.e., SARBO) and in 2003 for operations in the Gulf of Mexico (i.e., GRBO). On March 27, 2020, we completed a new consultation for SARBO (NMFS 2020a). This Opinion concluded the proposed action is not likely to jeopardize the continued existence of the following species or DPSs: NA or SA DPS of green sea turtle; Kemp's ridley, leatherback, or the NWA DPS of loggerhead sea turtle; the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, or SA DPSs of Atlantic sturgeon; shortnose sturgeon; giant manta ray; the U.S. DPS of smalltooth sawfish; Johnson's seagrass; or elkhorn, staghorn, lobed star, mountainous star, or boulder star coral. An ITS was issued for these affected species, which relied on running triennial take limits for all species aside from lethal take for smalltooth sawfish (9-year period) and corals (10-year period). We revised the GRBO in 2007 (NMFS 2007c), which concluded that: 1) Gulf of Mexico hopper dredging would adversely affect Gulf sturgeon and 4 sea turtle species (i.e., green, hawksbill, Kemp's ridley, and loggerheads) but would not jeopardize their continued existence; and 2) dredging in the Gulf of Mexico would not adversely affect leatherback sea turtles, smalltooth sawfish, or ESA-listed large whales. An ITS for adversely affected species was issued in this revised Opinion.

The above-listed regional Opinions consider maintenance dredging and sand mining operations. We have produced numerous other "free-standing" Opinions that analyzed hopper dredging projects (e.g., navigation channel improvements and beach restoration projects) that did not fall partially or entirely under the scope of actions contemplated by these regional Opinions. Any free-standing Opinions had its own ITS and determined that hopper dredging during the proposed action would not adversely affect any species of sea turtles or other listed species, or destroy or adversely modify critical habitat of any listed species.

4.2.1.3 Federal Vessel Activity

Watercraft are the greatest contributors to overall noise in the sea and have the potential to interact with sea turtles and giant manta ray, and to a much lesser extent, sturgeon, though direct impacts or propellers. Sound levels and tones produced are generally related to vessel size and speed. Larger vessels generally emit more sound than smaller vessels, and vessels underway with a full load, or those pushing or towing a load, are noisier than unladen vessels. Vessels operating at high speeds have the potential to strike sea turtles and giant manta ray. Potential sources of adverse effects from federal vessel operations in the action area include operations of the Bureau of Ocean Energy Management (BOEM), Federal Energy Regulatory Commission, USCG, NOAA, and USACE.

4.2.1.4 Military Activities

Military ordnance detonation also affects listed species, though the degree to which listed species are affected is largely unknown, though we do believe these activities may adversely affect sea turtles in particular. Section 7 consultations were conducted for U.S. Navy, U.S. Air Force, USCG, and U.S. Marine Corps activities.

4.2.1.5 Offshore Energy

Federal and state oil and gas exploration, production, and development are expected to result in some sublethal effects to protected species, including impacts associated with the explosive removal of offshore structures, seismic exploration, marine debris, and oil spills. Many Section 7 consultations have been completed on BOEM oil and gas lease activities. Until 2002, these Opinions concluded only 1 sea turtle take may occur annually due to vessel strikes. Through the Section 7 process, where applicable, we have and will continue to establish conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species. Subsequent Opinions (e.g., NMFS 2007d) have concluded that sea turtle takes may also result from marine debris and oil spills.

Impact of DWH Oil Spill on Status of Sea Turtles
On April 20, 2010, while working on an exploratory well approximately 50 nm offshore Louisiana, the semi-submersible drilling rig 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 finally capped on July 15, 2010. Millions of barrels of oil were released into the Gulf. 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 sea turtles.

At this time, the total effects of the oil spill on species found throughout the Gulf of Mexico, including ESA-listed sea turtles, are not known. Potential DWH-related impacts to all sea turtle species 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, loss of foraging resources which could lead to compromised growth and/or reproductive potential, harm to foraging, resting and/or nesting habitats, and disruption of nesting turtles and nests. Consequently, other than some emergency restoration efforts, monitoring actions (e.g., enhanced Gear Monitoring Team and stranding network coverage), and enforcement, most restoration efforts that occur pursuant to the Oil Pollution Act have yet to be determined and implemented, and so the ultimate restoration impacts on the species are unknowable at this time. State resource agencies have recently initiated several other restoration projects, though it will take some time until tangible results can be resolved.

During the response phase to the DWH oil spill (April 26-October 20, 2010) a total of 1,146 sea turtles were recovered, either as strandings (dead or debilitated generally onshore or nearshore) or were collected offshore during sea turtle search and rescue operations. Subsequent to the response phase a few sea turtles with visible evidence of oiling have been recovered as strandings. The available data on sea turtle strandings and response collections during the time of the spill are expected to represent a fraction (currently unknown) of the actual losses to the species, as most individuals likely were not recovered. The number of strandings does not provide insights into potential sublethal impacts that could reduce long-term survival or fecundity of individuals affected. It does, however, provide some insight into the potential relative scope of the impact among the sea turtle species in the area. Kemp's ridley sea turtles may have been the most affected sea turtle species, as they accounted for almost 71% of all recovered turtles (alive and dead), and 79% of all dead turtles recovered. Green turtles accounted for 17.5% of all recoveries (alive and dead), and 4.8% of the dead turtles recovered. Loggerheads comprised 7.7% of total recoveries (alive and dead) and 11% of the dead turtle recovered. The remaining turtles were hawksbills and decomposed hardshell turtles that were not identified to species. No leatherbacks were among the sea turtles recovered in the spill response area (note: leatherbacks were documented in the spill area, but they were not recovered alive or dead).

Although extraordinarily high numbers of threatened and endangered sea turtles were documented stranded (primarily within Mississippi Sound), during the DWH oil spill the vast majority of sea turtles recovered by the stranding network have shown no visible signs of oil. The DWH oil spill event increased awareness and human presence in the northern Gulf of Mexico, which likely resulted in some of the increased reporting of stranded turtles to the stranding network; however, we do not believe this factor fully explains the increases observed in 2010. We believe some of the increases in strandings may have been attributed to bycatch mortality in the shrimp fishery. As a result, on August 16, 2010, we reinitiated Section 7 consultation on Southeast state and federal shrimp fisheries based on a high level of strandings, elevated nearshore sea turtle abundance as measured by trawl catch per unit of effort, and lack of compliance with TED requirements. These factors indicated sea turtles may be affected by shrimp trawling to an extent not previously considered in the 2002 Opinion on the shrimp fisheries.

Another period of high stranding levels occurred in 2011, similar to that in 2010. We initiated investigations, including necropsies, to attempt to determine the cause of those strandings. Based on the findings, the 2 primary considerations for the cause of death of the turtles that were necropsied are forced submergence or acute toxicosis. With regard to acute toxicosis, sea turtle tissue samples were tested for biotoxins of concern in the northern Gulf of Mexico. Environmental information did not indicate a HAB of threat to marine animal health was present in the area. With regard to forced submergence, the only known plausible cause of forced submergence that could explain this event is incidental capture in fishing gear. We have assembled information regarding fisheries operating in the area during and just prior to these strandings. While there is some indication that lack of compliance with existing TED regulations and the operations of other trawl fisheries that do not require TEDs may have occurred in the area at the time of the strandings, direct evidence that those events caused the unusual level of strandings is not available. More information on the stranding event, including number of strandings, locations, and species affected, can be found at https://www.fisheries.noaa.gov/national/marine-life-distress/deepwater-horizon-oil-spill-2010-sea-turtles-dolphins-and-whales.

In addition to effects on subadult and adult sea turtles, the 2010 May through September sea turtle nesting season in the northern Gulf may also have been adversely affected by the DWH oil spill. Setting booms to protect beaches, cleanup activities, lights, people, and equipment all may have had unintended effects, such as preventing females from reaching nesting beaches and thereby reducing nesting in the northern Gulf of Mexico.

The oil spill may also have adversely affected emergence success. In the northern Gulf of Mexico area, approximately 700 nests are laid annually in the Florida Panhandle and up to 80 nests are laid annually in Alabama. Most nests are made by loggerhead sea turtles; however, a few Kemp's ridley and green turtle nests were also documented in 2010. Hatchlings begin emerging from nests in early to mid-July; the number of hatchlings estimated to be produced from northern Gulf sea turtle nests in 2010 was 50,000. To try to avoid the loss of most, if not all, of 2010's northern Gulf of Mexico hatchling cohort, all sea turtle nests laid along the northern Gulf Coast were visibly marked to ensure that nests were not harmed during oil spill cleanup operations that are undertaken on beaches. In addition, a sea turtle late-term nest collection and hatchling release plan was implemented to provide the best possible protection for sea turtle hatchlings emerging from nests in Alabama and the Florida Panhandle. Starting in June, northern Gulf of Mexico nests were relocated to the Atlantic to provide the highest probability of reducing the anticipated risks to hatchlings as a result of the DWH oil spill. A total of 274 nests, all loggerheads except for 4 green turtle and 5 Kemp's ridley nests, were translocated just prior to emergence from northern Gulf of Mexico beaches to the east coast of Florida so that the hatchlings could be released in areas not affected by the oil spill (Table 10). In mid-August, it was determined that the risks to hatchlings emerging from beaches and entering waters off the northern Gulf Coast had diminished significantly, and all nest translocations were ceased by August 19, 2010.

Table 10. Number of Turtle Nests Translocated from the Gulf Coast and Hatchlings Released in the Atlantic Ocean. The sea turtle nest translocation effort ceased on August 19, 2010.

Turtle Species	Translocated Nests	Hatchlings Released
Green turtle (*Chelonia mydas*)	4	455
Kemp's ridley turtle (*Lepidochelys kempii*)	5	125
Loggerhead turtle (*Caretta caretta*)	265[1]	14,216

[1] Does not include 1 nest that included a single hatchling and no eggs.

The survivorship and future nesting success of individuals from one nesting beach being transported to and released at another nesting beach is unknown. The loggerheads nesting and emerging from nests in the Florida Panhandle and Alabama are part of the NGMRU and differ genetically from loggerheads produced along the Atlantic Coast of Florida, but they are part of NWA DPS. Evidence suggests that some portion of loggerheads produced on Northern Gulf beaches are transported naturally into the Atlantic by currents and spend portions of their life cycles away from the Gulf of Mexico. This is based on the presence of some loggerheads with a northern Gulf of Mexico genetic signature in the Atlantic. These turtles are assumed to make their way back to the Gulf of Mexico as subadults and adults. It is unknown what the impact of the nesting relocation efforts will be on the NGMRU in particular, or the NWA DPS generally.

Loggerhead nesting in the northern Gulf of Mexico represents a small proportion of overall Florida loggerhead nesting and an even smaller proportion of the NWA DPS. The 5-year average (2006-2010) for the statewide number of loggerhead nests in the state of Florida is 56,483 nests annually (FWC nesting database) versus an average of well under 1,000 nests per year for the northern Gulf of Mexico (approximately 700 in 2010). We do not know what the impact of relocating 265 nests will be on the 2010 nesting cohort compared to the total of approximately 700 nests laid on Northern Gulf of Mexico beaches. While there may be a risk of possible increased gene flow across loggerhead recovery units, all are within the NWA DPS and would likely not be on a scale of conservation concern. Recovery units are subunits of the listed species that are geographically or otherwise identifiable and essential to the recovery of the species. Recovery units are individually necessary to conserve genetic robustness, demographic robustness, important life history stages, or some other feature necessary for long-term sustainability of the species. Recovery units are not necessarily self-sustaining viable units on their own, but instead need to be collectively recovered to ensure recovery of the entire listed entity. Recovery criteria must be met for all recovery units identified in the Recovery Plan before the NWA DPS can be considered for delisting.

As noted earlier, the vast majority of sea turtles collected in relation to the DWH oil spill event were Kemp's ridleys; 328 were recovered alive and 481 were recovered dead. We expect that additional mortalities occurred that were undetected and are, therefore, currently unknown. It is likely that the Kemp's ridley sea turtle was also the species most impacted by the DWH spill event on a population level. Relative to the other species, Kemp's ridley populations are much smaller, yet recoveries during the DWH oil spill response were much higher. The location and timing of the DWH oil spill event were also important factors. Although significant assemblages of juvenile Kemp's ridleys occur along the U.S. Atlantic coast, Kemp's ridley sea turtles use the Gulf of Mexico as their primary habitat for most life stages, including all of the mating and nesting. As a result, all mating and nesting adults in the population necessarily spend significant time in the Gulf of Mexico, as do all hatchlings as they leave the beach and enter the pelagic environment. Still, not all of those individuals will have encountered oil and/or dispersants, depending on the timing and location of their movements relative to the location of the subsurface and surface oil. In addition to mortalities, the effects of the spill may have included disruptions to foraging and resource availability, migrations, and other unknown effects as the spill began in late April just before peak mating/nesting season (May-July) although the distance from the DWH well to the primary mating and nesting areas in Tamaulipas, Mexico greatly reduces the chance of these disruptions to adults breeding in 2010. Yet, turtle returns from nesting beaches to foraging areas in the northern Gulf of Mexico occurred while the well was still spilling oil. At this time, we cannot determine the specific reasons accounting for year-to-year fluctuations in numbers of Kemp's ridley nests (the number of nests increased in 2011 as compared to 2010); however, there may yet be long-term population impacts resulting from the oil spill. How quickly the species returns to the previous fast pace of recovery may depend in part on how much of an impact the DWH event has had on Kemp's ridley food resources (Crowder and Heppell 2011).

Eighty-eight loggerhead sea turtles have been documented within the designated spill area as part of the response efforts; 67 were dead and 21 were alive. It is unclear how many of those without direct evidence of oil were actually impacted by the spill and spill-related activities versus other sources of mortality. There were likely additional mortalities that were undetected and, therefore, currently unknown. Although we believe that the DWH event had adverse effects on loggerheads, the population level effect was not likely as severe as it was for Kemp's ridleys. In comparison to Kemp's ridleys, we believe the relative proportion of the population exposed to the effects of the event was much smaller, the number of turtles recovered (alive and dead) are fewer in absolute numbers, and the overall population size is believed to be many times larger. Additionally, unlike Kemp's ridleys, the majority of nesting for the NWA DPS occurs on the Atlantic coast. 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 because of impacts to nesting (as described above) and a larger proportion of the NGMRU recovery unit, especially mating and nesting adults, being exposed to the spill. However, the impacts to that recovery unit, and the possible effect of such a disproportionate impact on that small recovery unit to the NWA DPS and the species, remain unknown.

Green sea turtles comprised the second-most common species recovered as part of the DWH response. Of the 201 green turtles recovered 29 were found dead or later died while undergoing rehabilitation. The mortality number is lower than that for loggerheads despite loggerheads having far fewer total strandings, but this is because the majority of green turtles came from the offshore rescue (pelagic stage), of which almost all (of all species) survived after rescue, whereas a greater proportion of the loggerhead recoveries were nearshore neritic stage individuals found dead. While green turtles regularly use the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic. As described in the Status of the Species section, nesting is relatively rare on the northern Gulf Coast. Similar to loggerhead sea turtles, it is expected that adverse impacts have occurred, but that the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event is relatively low. Thus, the population-level impact is likely much smaller than for Kemp's ridleys.

Assessing the current impacts of this oil spill on Gulf sturgeon and their designated critical habitat is difficult because so much remains unknown or unclear about the impacts to the environment and habitat. Given these uncertainties, it is not practical to speculate on spill effects to the Gulf sturgeon environmental baseline at this time; however, we expect the primary route of effects to designated critical habitat from the release of oil and subsequent cleanup efforts is to the benthos and the benthic community it supports. There are at least 2 routes of exposure: 1) the suffocation of infaunal organisms; and 2) toxicity of substrate. Both of these effects would impact the abundance of Gulf sturgeon prey. The long-term impact to Gulf sturgeon and their designated critical habitat from exposure to oil and the subsequent response and clean-up efforts is currently unknown.

4.2.1.6 Construction and Operation of USACE-Permitted Fishing Piers

We have consulted with the USACE on the construction and operation of a number of fishing piers that may have adverse effects to sea turtles, smalltooth sawfish, and sturgeon because of the potential impacts of recreational fishing from these piers on these species. For instance, from 2010-2015, 19 fishing piers in Mississippi documented a total of 863 sea turtle captures; the Washington Street Pier in Bay St. Louis documented the most captures, with 254 sea turtles documented. Any consultation that concluded the respective pier would adversely affect listed species, also concluded the action would not jeopardize their continued existence, and an ITS for adversely affected species was issued in each Opinion.

4.2.1.7 Federally-Permitted Discharges

Federally regulated stormwater and industrial discharges and chemically treated discharges from sewage treatment systems may impact Gulf sturgeon and their critical habitat. We continue to consult with EPA to minimize the effects of these activities on both listed species and designated critical habitat. In addition, other federally permitted construction activities, such as beach restoration, have the potential to impact Gulf sturgeon critical habitat.

4.2.1.8 ESA Permits

Regulations developed under the ESA allow for the issuance of permits allowing take of certain ESA-listed species for the purposes of scientific research under Section 10(a)(1)(a) of the ESA. Since issuance of the permit is a federal activity, the action must be reviewed for compliance with Section 7(a)(2) of the ESA to ensure that issuance of the permit does not result in jeopardy to the species or adverse modification of its critical habitat. Authorized activities range from photographing, weighing, and tagging sea turtles incidentally taken in fisheries, to blood sampling, tissue sampling (biopsy), and performing laparoscopy on intentionally captured sea turtles. 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 (and are) nonlethal. Section 10 research is also conducted on smalltooth sawfish, which may include capturing, handling, collection of tissue samples, and tagging smalltooth sawfish in Florida waters (both South Atlantic and Gulf of Mexico). To date, we have issued 4 permits for directed research on smalltooth sawfish; all smalltooth sawfish take authorized under these permits is nonlethal.

There are no federal permits for Gulf sturgeon research. The states have permitting authority (56 FR 49653; September 30, 1991) and no annual reporting is required. We and USFWS established a standardized sampling protocol with the Gulf sturgeon researchers in 2010. Procedures for tagging were established, PIT tag frequencies were standardized, and a common datasheet was established. Tag information and morphometric data are being stored in a shared database managed by us. A similar workshop to discuss and establish monitoring protocols occurred in 2012. There are currently 3 Section 10(a)(1)(A) scientific research permits issued to study Atlantic sturgeon. These studies authorize researchers to anesthetize, collect eggs, attach external instrument (e.g., satellite tags), insert internal instrument (e.g., sonic tags), mark/PIT tag, measure, photograph/video, fin clip, and weigh animals. Most takes authorized under these permits are expected to be nonlethal, but there are a few anticipated mortalities. As with other ESA-listed species research permitting, since issuance of these permits is a federal activity, the action must be reviewed for compliance with Section 7(a)(2) of the ESA to ensure that issuance of the permit does not result in jeopardy to the species.

4.2.2 State or Private Actions

A number of activities in state waters that may directly or indirectly affect listed species include recreational and commercial fishing, construction, discharges from wastewater systems, dredging, ocean pumping and disposal, and aquaculture facilities. The impacts from some of these activities are difficult to measure. However, where possible, conservation actions through the ESA Section 7 process, ESA Section 10 permitting, and state permitting programs are being implemented to monitor or study impacts from these sources. Increasing coastal development and ongoing beach erosion will result in increased demands by coastal communities, especially beach resort towns, for periodic privately funded or federally sponsored beach nourishment projects. Some of these activities may affect listed species (e.g., sea turtles and Gulf sturgeon) and their critical habitat by burying nearshore habitats that serve as foraging areas. Additional discussion on some of these activities follows.

4.2.2.1 State Fisheries

Various fishing methods used in state commercial and recreational fisheries, including gillnets, fly nets, trawling, pot fisheries, pound nets, and vertical line are all known to incidentally take sea turtles, but information on these fisheries is sparse (NMFS 2001). Most of the state data are based on extremely low observer coverage, or sea turtles were not part of data collection; thus, these data provide insight into gear interactions that could occur but are not indicative of the magnitude of the overall problem.

State fisheries conducted in waters off the coast of Florida are known to occasionally capture smalltooth sawfish. Fishers who capture smalltooth sawfish most commonly are recreationally fishing for snook (*Centropomus undecimalis*), redfish (*Scianops ocellatus*), and sharks (Simpfendorfer and Wiley 2004). Encounter data indicate that the majority of these interactions are nonlethal. We are encouraging the FWC to apply for an ESA Section 10 incidental take permit for its fisheries.

Both Atlantic and Gulf sturgeon are also known to be adversely affected by gillnets (e.g., Georgia commercial shad fisheries) in state waters. In fact, given this gear type is used most frequently in state waters, state fisheries may have a greater impact on sturgeon than federal fisheries using this same gear type. In Georgia, the commercial shad fisheries incidentally capture Atlantic sturgeon. The Gulf sturgeon recovery plan (NMFS and USFWS 1995) documents that Gulf sturgeon are occasionally incidentally captured in state fisheries in bays and sounds along the northern Gulf of Mexico). In the Pearl River (i.e., along the Mississippi/Louisiana border) a trammel/gillnet fishery is conducted for gar. Because of the gear (minimum of 3-in square mesh, up to 3,000 ft in length) and the year-round nature of the fishery, it is probable that Gulf sturgeon are intercepted in this fishery. While state regulations prohibit taking or possession of whole or any body parts, including roe, there is no reporting to determine capture or release rates.

Trawl Fisheries
Other trawl fisheries, such as ones operating for blue crab and sheepshead, may also interact with sea turtles and sturgeon in state waters. Many of these vessels are shrimp trawlers that alter their gear in other times of the year to target these other species. At this time, however, we lack sufficient information to quantify the level of anticipated take that may be occurring in these other trawl fisheries.

Recreational Fishing
Recreational fishing from private vessels may occur in the action area, and these activities may interact with sea turtles, smalltooth sawfish, sturgeon, and giant manta ray. For example, observations of state recreational fisheries have shown that loggerhead sea turtles are known to bite baited hooks and frequently ingest the hooks. Hooked turtles have been reported by the public fishing from boats, fishing piers (see previous discussion in Section 4.2.1.5), and beach, banks, and jetties and from commercial anglers fishing for reef fish and for sharks with both single rigs and bottom longlines. Additionally, lost fishing gear such as line cut after snagging on rocks, or discarded hooks and line, can also pose an entanglement threat to sea turtles in the area. A detailed summary of the known impacts of hook-and-line incidental captures to loggerhead sea turtles can be found in the SEFSC Turtle Expert Working Group (TEWG) reports (TEWG 1998; TEWG 2000).

4.2.2.2 Vessel Traffic

Commercial traffic and recreational boating pursuits can have adverse effects on sea turtles and giant manta ray in particular via propeller and boat strike damage. The STSSN includes many records of vessel interactions (propeller injury) with sea turtles, and giant manta ray are also frequently observed with prop scars on their dorsal surface. Data show that vessel traffic is one cause of sea turtle mortality (Hazel and Gyuris 2006; Lutcavage et al. 1997). Stranding data show that vessel-related injuries are noted in stranded sea turtles.[13] Data indicate that live- and dead-stranded sea turtles showing signs of vessel-related injuries continue in a high percentage of stranded sea turtles in coastal regions of the southeastern United States, particularly off Florida where there are high levels of vessel traffic.

4.2.2.3 Coastal Development

Beachfront development, lighting, and beach erosion control all are ongoing activities along the southeastern U.S. coastline (i.e., throughout the action area). These activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. Still, more and more coastal counties are adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting.

4.2.3 Other Potential Sources of Impacts to the Environmental Baseline

4.2.3.1 Stochastic events

Stochastic (i.e., random) events, such as hurricanes, occur in the southeastern U.S., and can affect the action area. These events are by nature unpredictable, and their effect on the recovery of the species is unknown; yet, they have the potential to directly impede recovery if animals die as a result or indirectly if important habitats are damaged. Conversely, these events, such as the record 2020 Atlantic hurricane season, may also result in some benefits to listed species, particularly sea turtles. For example, the impacts of hurricanes may compromise fisheries infrastructure and reduce fishing effort, which may subsequently reduce fishery related bycatch. Other stochastic events, such as a winter cold snap, can injure or kill sea turtles.

4.2.3.2 Marine Pollution and Environmental Contamination

In general, marine pollution includes a wide variety of impacts stemming from a diversity of activities and sources. Sources of pollutants within or adjacent to the action area include, but are not limited to, marine debris and plastics, noise pollution from vessel traffic and military training activities, atmospheric loading of pollutants such as PCBs, agricultural and industrial runoff into rivers and canals emptying into bays and the ocean (e.g., Mississippi River into the Gulf of Mexico), and groundwater and other discharges. Nutrient loading from land-based sources such as coastal community discharges is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effects on larger embayments are unknown. An example is the large area of the Louisiana continental shelf with seasonally-depleted oxygen levels (< 2 mg/Liter) is caused by eutrophication from both point and non-point sources. Most aquatic species cannot survive at such low oxygen levels and these areas are known as "dead zones." The oxygen depletion, referred to as hypoxia, begins in late spring, reaches a maximum in mid-summer, and

[13] https://www.fisheries.noaa.gov/national/marine-life-distress/sea-turtle-stranding-and-salvage-network

disappears in the fall. Since 1993, the average extent of mid-summer, bottom-water hypoxia in the northern Gulf of Mexico has been approximately 16,000 km^2, approximately twice the average size measured between 1985 and 1992. The hypoxic zone attained a maximum measured extent in 2002, when it was about 22,000 km^2, which is larger than the state of Massachusetts (USGS 2008). The 2020 Gulf of Mexico hypoxic zone measured 5,480 km^2 and was the 3rd smallest in the 34-year record of surveys; the 5-year average is now down to 14,007 km^2 (EPA 2020). The hypoxic zone has impacts on the animals found there, including sea turtles, and the ecosystem-level impacts continue to be investigated.

Additional direct and indirect sources of pollution include dredging (i.e., resuspension of pollutants in contaminated sediments), aquaculture, and oil and gas exploration and extraction, each of which can degrade marine habitats used by sea turtles (Colburn et al. 1996) and other listed species. The development of marinas and docks in inshore waters can negatively impact nearshore habitats. An increase in the number of docks built increases boat and vessel traffic. Fueling facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive estuarine and coastal habitats. Although these contaminant concentrations do not likely affect the more pelagic waters, the species of turtles analyzed in this Opinion travel between near shore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles.

Sea turtles may ingest marine debris, particularly plastics, which can cause intestinal blockage and internal injury, dietary dilution, malnutrition, and increased buoyancy, which, in turn, can result in poor health, reduced growth rates and reproductive output, or death (Nelms et al. 2016). Entanglement in plastic debris (including ghost fishing gear) is known to cause lacerations, increased drag—which reduces the ability to forage effectively or escape threats—and may lead to drowning or death by starvation. While more widely documented in sea turtles, entanglement in marine debris has also been noted for giant manta ray and smalltooth sawfish.

The Gulf of Mexico is an area of high-density offshore oil extraction with chronic, low-level spills and occasional massive spills (e.g., DWH oil spill event). Oil spills can impact wildlife directly through 3 primary pathways: 1) ingestion—when animals swallow oil particles directly or consume prey items that have been exposed to oil; 2) absorption—when animals come into direct contact with oil; and 3) inhalation—when animals breath volatile organics released from oil or from "dispersants" applied by response teams in an effort to increase the rate of degradation of the oil in seawater. Several aspects of sea turtle biology and behavior place them at particular risk, including the lack of avoidance behavior, indiscriminate feeding in convergence zones, and large pre-dive inhalations (Milton et al. 2003). When large quantities of oil enter a body of water, chronic effects such as cancer, and direct mortality of wildlife becomes more likely (Lutcavage et al. 1997). Oil spills in the vicinity of nesting beaches just prior to or during the nesting season could place nesting females, incubating egg clutches, and hatchlings at significant risk (Fritts et al. 1982; Lutcavage et al. 1997; Witherington 1999). Continuous low-level exposure to oil in the form of tar balls, slicks, or elevated background concentrations also challenge animals facing other natural and anthropogenic stresses. Types of trauma can include skin irritation, altering of the immune system, reproductive or developmental damage, and liver disease (Keller et al. 2004; Keller et al. 2006). Chronic exposure may not be lethal by itself, but it may impair a turtle's overall fitness so that it is less able to withstand other stressors (Milton et al. 2003).

The earlier life stages of living marine resources are usually at greater risk from an oil spill than adults. This is especially true for sea turtle hatchlings, since they spend a greater portion of their time at the sea surface than adults; thus, their risk of exposure to floating oil slicks is increased (Lutcavage et al. 1995). One of the reasons might be the simple effects of scale: for example, a given amount of oil may overwhelm a smaller immature organism relative to the larger adult. The metabolic machinery an animal uses to detoxify or cleanse itself of a contaminant may not be fully developed in younger life stages. Also, in early life stages, animals may contain proportionally higher concentrations of lipids, to which many contaminants such as petroleum hydrocarbons bind. Most reports of oiled hatchlings originate from convergence zones, ocean areas where currents meet to form collection points for material at or near the surface of the water.

Unfortunately, little is known about the effects of dispersants on sea turtles, and such impacts are difficult to predict in the absence of direct testing. While inhaling petroleum vapors can irritate turtles' lungs, dispersants can interfere with lung function through their surfactant (detergent) effect. Dispersant components absorbed through the lungs or gut may affect multiple organ systems, interfering with digestion, respiration, excretion, and/or salt-gland function—similar to the empirically demonstrated effects of oil alone (Shigenaka et al. 2003). Oil cleanup activities can also be harmful. Earth-moving equipment can dissuade females from nesting and destroy nests, containment booms can entrap hatchlings, and lighting from nighttime activities can misdirect turtles (Witherington 1999).

There are studies on organic contaminants and trace metal accumulation in green and leatherback 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 turtles 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. 2008). It is thought that dietary preferences were likely to be the main differentiating factor among species. Decreasing lipid contaminant burdens with turtle size were observed in green turtles, most likely attributable to a change in diet with age. Sakai et al. (1995) found the presence of metal residues points for material at or near the surface of the water. Sixty-five of 103 post-hatchling loggerheads in convergence zones off Florida's east coast were found with tar in the mouth, esophagus or stomach (Loehefener et al. 1989). Thirty-four percent of post-hatchlings captured in *Sargassum* off the Florida coast had tar in the mouth or esophagus and more than 50% had tar caked in their jaws (Witherington 1994). These zones aggregate oil slicks, such as a Langmuir cell, where surface currents collide before pushing down and around, and represents a virtually closed system where a smaller weaker sea turtle can easily become trapped (Carr 1987; Witherington 2002). Lutz and Lutcavage (1989) reported that hatchlings have been found apparently starved to death, their beaks and esophagi blocked with tarballs. Hatchlings sticky with oil residue may have a more difficult time crawling and swimming, rendering them more vulnerable to predation.

Frazier (1980) suggested that olfactory impairment from chemical contamination could represent a substantial indirect effect in sea turtles, since a keen sense of smell apparently plays an important role in navigation and orientation. A related problem is the possibility that an oil spill impacting nesting beaches may affect the locational imprinting of hatchlings, and thus impair their ability to return to their natal beaches to breed and nest (Milton et al. 2003). Whether hatchlings, juveniles, or adults, tar balls in a turtle's gut are likely to have a variety of effects – starvation from gut blockage, decreased absorption efficiency, absorption of toxins, effects of general intestinal blockage (such as local necrosis or ulceration), interference with fat metabolism, and buoyancy problems caused by the buildup of fermentation gases (floating prevents turtles from feeding and increases their vulnerability to predators and boats), among others. Also, trapped oil can kill the seagrass beds that turtles feed upon.

Pollution from industrial, agricultural, and municipal activities is believed responsible for a suite of physical, behavioral, and physiological impacts to sturgeon worldwide (Agusa et al. 2004; Barannikova 1995; Bickham et al. 1998; Billard and Lecointre 2000; Kajiwara 2003; Karpinsky 1992; Khodorevskaya et al. 1997; Khodorevskaya and Krasikov 1999). Several characteristics of Gulf sturgeon (i.e., long lifespan, extended residence in riverine and estuarine habitats, benthic predator) predispose the species to long-term and repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants. Chemicals and metals such as chlordane, dichlorodiphenyldichloroethylene, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later incorporated into the food web as they are consumed by benthic feeders, such as sturgeon or macroinvertebrates. Some of these compounds may affect physiological processes and impede the ability of a fish to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing DO, altering pH, and altering other water quality properties.

While laboratory results are not available for Gulf sturgeon, signs of stress observed in shortnose sturgeon exposed to low DO included reduced swimming and feeding activity coupled with increased ventilation frequency (Campbell and Goodman 2004). Niklitschek (2001) observed that egestion levels for Atlantic and shortnose sturgeon juveniles increased significantly under hypoxia, indicating that consumed food was incompletely digested. Behavioral studies indicate that Atlantic and shortnose sturgeon are quite sensitive to ambient conditions of oxygen and temperature: in choice experiments juvenile sturgeons consistently selected normoxic (normal oxygen level) over hypoxic (low oxygen level) conditions (Niklitschek 2001). Beyond escape or avoidance, sturgeons respond to hypoxia through increased ventilation, increased surfacing (to ventilate relatively oxygen-rich surficial water), and decreased swimming and routine metabolism (Crocker and Cech Jr. 1997; Niklitschek 2001; Nonnotte et al. 1993; Secor and Gunderson 1998).

The majority of published data regarding contaminants and sturgeon health are limited to reports of tissue concentration levels. While these data are useful and allow for comparison between individuals, species, and regions, they do not allow researchers to understand the impacts of the concentrations. There is expectation that Gulf sturgeon are being negatively impacted by organic and inorganic pollutants given high concentration levels (Berg 2006). Gulf sturgeon collected from a number of rivers between 1985 and 1991 were analyzed for pesticides and heavy metals (Bateman and Brim 1994); concentrations of arsenic, mercury, DDT metabolites, toxaphene, polycyclic aromatic hydrocarbons, and aliphatic hydrocarbons were sufficiently high to warrant concern. More recently, 20 juvenile Gulf sturgeon from the Suwannee River, Florida, exhibited an increase in metals concentrations with an increase in individual length (Alam et al. 2000).

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

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

The EPA published its second edition of the National Coastal Condition Report in 2004, which is a "report card" summarizing the status of coastal environments along the coast of the United States (EPA 2004). The report analyzes water quality, sediment, coastal habitat, benthos, and fish contaminant indices to determine status. In contrast to the Northeast (Virginia - Maine), which received an overall grade of F, the Southeast region (North Carolina - Florida) received an overall grade of B-, which is the best rating in the nation with no indices below a grade of C.

Areas of concern that had poor index scores within the action area include were Pamlico Sound and the ACE Basin for water quality, and St. Johns River for sediment. There was also a mixture of poor benthic scores scattered along Southeast region.

Smalltooth sawfish may be indirectly affected by anthropogenic marine pollution. As described in Section 3, no specific information is available on the effects of pollution on smalltooth sawfish, but evidence from other elasmobranchs suggests that pollution disrupts endocrine systems and potentially leads to reproductive failure (Gelsleichter et al. 2006). Smalltooth sawfish have been encountered with polyvinyl pipes and fishing gear on their rostrum (Gregg Poulakis, FWC, pers. comm. to Shelley Norton, NMFS, 2007).

4.2.4 Conservation and Recovery Actions Shaping the Environmental Baseline

We have 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 and Gulf of Mexico reef fish fisheries, and TED requirements for the Southeast shrimp trawl fisheries. These regulations have relieved some of the stressors on sea turtle populations. Other actions taken by states, have also benefited the recovery of listed species. For instance, regulations restricting the use of entangling nets (including gillnets, trammel nets, and purse seines) were banned in Florida state waters in 1994. Although intended to restore the populations of inshore gamefish, this action removed possibly the greatest source of fishing mortality on smalltooth sawfish (Simpfendorfer 2002).

Under Section 6 of the ESA, we may enter into cooperative research and conservation agreements with states to assist in recovery actions of listed species. We have agreements with all states in the action area for sea turtles. We have also established partnerships for cooperative research on Gulf sturgeon via conservation agreements in the Gulf of Mexico with the States of Florida, Alabama, Mississippi, and Louisiana. Prior to issuance of these agreements, the proposal must be reviewed for compliance with Section 7 of the ESA.

Along with cooperating states, we have established an extensive network of STSSN participants along the Atlantic and Gulf of Mexico coasts that not only collect data on dead sea turtles, but also rescue and rehabilitate any live stranded sea turtles. The network, which includes federal, state and private partners, encompasses the coastal areas of the 18-state region from Maine to Texas, and includes portions of the U.S. Caribbean. Data are compiled through the efforts of network participants who document marine turtle strandings in their respective areas and contribute those data to the centralized STSSN database

Research, monitoring, and outreach efforts on smalltooth sawfish are providing valuable information on which to base effective conservation management measures. Monitoring and research programs for the smalltooth sawfish are ongoing in southwest Florida. Surveys are conducted using longlines, setlines, gillnets, rod and reel, and seine nets. Cooperating fishers, guides, and researchers are also reporting smalltooth sawfish they encounter. Data collected are providing new insight on the species' current distribution, abundance, and habitat use patterns. Public outreach efforts help to educate the public on smalltooth sawfish status and proper handling techniques, which minimizes interaction, injury, and mortality of encountered smalltooth sawfish.

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

We 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 hard-shelled turtles caught in fishing or scientific research gear.

A final rule (70 FR 42508) published on July 25, 2005, allows any of our agents or employees, the USFWS, the U.S. Coast Guard, or any other federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, 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. We already afford the same protection to sea turtles listed as threatened under the ESA (50 CFR 223.206(b)).

TED regulations, which were first introduced in the 1990s in the shrimp fisheries (with a major subsequent revision in 2003), have benefited sea turtle populations, as well as other species like sturgeon by reducing incidental fisheries bycatch and mortality (ASSRT 2007). We have transferred TED technology to numerous foreign countries, and require an equivalent TED program in any foreign country wishing to import wild-caught shrimp into the U.S. via Section 609 of P.L. 101-162. We have also required TED use in the summer flounder fishery south of Cape Charles, Virginia, and are exploring TED designs for other fisheries (e.g., flynet fishery). Other gear-related modifications in other fisheries, such as the chain mat requirement in the scallop fishery, are also aimed at reducing overall fisheries bycatch mortality of sea turtles.

We have also published rules to require selected fishing vessels to carry observers to collect data on sea turtle interactions during fishing operations (August 3, 2007, 72 FR 43176), and to implement sea turtle release gear requirements and release protocols in specific commercial fisheries (e.g., South Atlantic snapper-grouper fishery, November 8, 2011, 76 FR 69230).

Other Actions
We helped to complete 5-year status reviews in 2007 for green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles. These reviews were conducted to comply with the ESA mandate for periodic status evaluation of listed species to ensure that their threatened or endangered listing status remains accurate. Each review determined that no delisting or reclassification of a species status (i.e., threatened or endangered) was warranted at this time. Further review of species data for the green, hawksbill, leatherback, and loggerhead sea turtles was recommended to evaluate whether DPSs should be established for these species (NMFS and USFWS 2007a; NMFS and USFWS 2007b; NMFS and USFWS 2007c; NMFS and USFWS 2007d; NMFS and USFWS 2007e). The Services completed a revised recovery plan for the loggerhead sea turtle on December 8, 2008 (NMFS and USFWS 2008) and published a final rule on September 22, 2011, listing loggerhead sea turtles as separate DPSs. A revised recovery plan for the Kemp's ridley sea turtle was completed on September 22, 2011. On October 10, 2012, we announced initiation of 5-year reviews of Kemp's ridley, olive ridley, leatherback, and hawksbill sea turtles, and requested submission of any pertinent information on those sea turtles that has become since their last status review in 2007.

In 2009, we completed a 5-year status review with USFWS for Gulf sturgeon (USFWS and NMFS 2009) and concluded that the species continues to meet the status of a threatened species. As part of that review, we also critiqued the recovery criteria listed in the 1995 Recovery Plan (USFWS and GSMFC 1995) and concluded that new criteria are necessary to: 1) reflect the best available and most up-to date information on the biology of the species; 2) address the 5 statutory listing/recovery factors; and 3) improve monitoring methods for demonstrating progress towards reducing threats and for determining when the protections of the ESA are no longer necessary. We are actively working to revise and update the 1995 Gulf Sturgeon Recovery Plan.

On January 21, 2009, we published the final recovery plan for the U.S. DPS of smalltooth sawfish. We are implementing recovery actions identified in the plan based on the recovery action's priority and available funding. Additionally, a 5-year review of the species status was published in October of 2010. The 5-year review concluded that the U.S. DPS of smalltooth sawfish remains vulnerable to extinction, and the species still meets the definition of endangered under the ESA, in that the species is in danger of extinction throughout its range. The recovery plan and the 5-year review are available at http://sero.nmfs.noaa.gov/pr/SmalltoothSawfish.htm.

4.3 Summary

In summary, several factors adversely affect sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish in the action area. These factors are ongoing and are expected to continue to occur contemporaneously with the proposed action. Fisheries in the action area likely had the greatest adverse impacts on sea turtles in the mid to late 1980s, when effort in most fisheries was near or at peak levels. With the decline of the health of managed species, effort since that time has generally been declining. Over the past 5 years, the impacts associated with fisheries have also been reduced through the Section 7 consultation process and regulations implementing effective bycatch reduction strategies. However, interactions with commercial and recreational fishing gear are still ongoing and are expected to continue to occur contemporaneously with the proposed action. Other environmental impacts including effects of vessel operations, additional military activities, dredging, oil and gas exploration, permits allowing take under the ESA, private vessel traffic, and marine pollution have also had and continue to have adverse effects on sea turtles and sturgeon in the action area in the past. The DWH oil spill is expected to have had an adverse impact on the baseline for sea turtles, but the extent of that impact is not yet well understood. Despite smalltooth sawfish being highly susceptible to entanglement, few interactions are reported or documented from the action area. Impacts on smalltooth sawfish over the last several decades may be limited in large part by the scarcity of smalltooth sawfish in the action area and due to lack of reporting. As the population slowly grows, fisheries and other activity stressors in the action area may have a greater impact on the species. While there is a paucity of information on impacts to giant manta ray, we expect ongoing and future research on the species will improve this deficit. Finally, actions to conserve and recover sea turtles, Atlantic and Gulf sturgeon, and smalltooth sawfish have significantly increased over the past 10 years and are expected to continue.

4.4 Climate Change

In addition to the information on climate change presented in the Section 3 (Status of the Species) for sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish the discussion below presents further background information on global climate change as well as past and predicted future effects of global climate change we expect throughout the action area. Also, below is the available information on predicted effects of climate change in the action area and how listed sea turtles and fish species may be affected by those predicted environmental changes. The effects are summarized on the time span of the proposed action, for which we can realistically analyze impacts, yet are discussed and considered for longer time periods when feasible. Yet, as mentioned previously, the potential effects, and the expected related climate change effects to ESA-listed species, are the result of slow and steady shift or alterations over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As a result, for the purposes of this Opinion we have elected to view the effects of climate change on affected species on a more manageable and predictable 10-year time period due to this reality. While climate change is also relevant to the Cumulative Effects section of this Opinion, we are synthesizing all additional information here rather than include partial discussions in other sections of this Opinion.

Background Information on Global Climate Change
The global mean temperature has risen 0.76ºC (1.36°F) over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007). Precipitation has increased nationally by 5%-10%, mostly due to an increase in heavy downpours (NAST 2000). In comparison, ocean temperatures have only increased by about 0.18°F in the last century, with the changes occurring from the surface to depths of about 2,300 ft. There is a high confidence, based on substantial new evidence, that observed changes in marine systems are

associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and other pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007); these trends are most apparent over the past few decades. Information on future impacts of climate change in the action area is discussed below.

Climate model projections exhibit a wide range of plausible scenarios for both temperature and precipitation over the next century. Both of the principal climate models used by the National Assessment Synthesis Team (NAST) project warming in the southeast by the 2090s, but at different rates (NAST 2000): the Canadian model scenario shows the southeast U.S. experiencing a high degree of warming, which translates into lower soil moisture as higher temperatures increase evaporation; the Hadley model scenario projects less warming and a significant increase in precipitation (about 20%). The scenarios examined, which assume no major interventions to reduce continued growth of world greenhouse gases (GHG), indicate that temperatures in the U.S. will rise by about 5°-9°F on average in the next 100 years, which is more than the projected global increase (NAST 2000). A warming of about 0.4°F per decade is projected for the next 2 decades over a range of emission scenarios (IPCC 2007). This temperature increase will very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. Climate warming has resulted in increased precipitation, river discharge, and glacial and sea-ice melting (Greene et al. 2008).

The past 3 decades have witnessed major changes in ocean circulation patterns in the Arctic, and these were accompanied by climate associated changes as well (Greene et al. 2008). Shifts in atmospheric conditions have altered Arctic Ocean circulation patterns and the export of freshwater to the North Atlantic (Greene et al. 2008, IPCC 2007). With respect specifically to the North Atlantic Oscillation (NAO), changes in salinity and temperature are expected to be the result of changes in the earth's atmosphere caused by anthropogenic forces (IPCC 2007). The NAO impacts climate variability throughout the Northern Hemisphere (IPCC 2007). Data from the 1960s through 2006 show that the NAO index increased from minimum values in the 1960s to strongly positive index values in the 1990s, but declined since (IPCC 2007). This warming extends more than 0.62 miles deep—deeper than anywhere in the world oceans—and is particularly evident under the Gulf Stream/ North Atlantic Current system (IPCC 2007). On a global scale, large discharges of freshwater into the North Atlantic subarctic seas can lead to intense stratification of the upper water column and a disruption of North Atlantic Deepwater (NADW) formation (Greene et al. 2008; IPCC 2007). There is evidence that the NADW has already freshened significantly (IPCC 2007). This in turn can lead to a slowing down of the global ocean thermohaline (large-scale circulation in the ocean that transforms low-density upper ocean waters to higher density intermediate and deep waters and returns those waters back to the upper ocean), which can have climatic ramifications for the whole earth system (Greene et al. 2008).

While predictions are available regarding potential effects of climate change globally, it is more difficult to assess the potential effects of climate change over the next few decades on smaller geographic scales, such as the Mississippi Sound or the Mid-Atlantic Bight, especially as climate variability is a dominant factor in shaping coastal and marine systems. The effects of future change will vary greatly in diverse coastal regions for the U.S. Warming is very likely to continue in the U.S. over the next 25 to 50 years regardless of reduction in GHG emissions due to emissions that have already occurred (NAST 2000); therefore, it is also expected to continue during the operation of the shrimp fisheries. It is very likely that the magnitude and frequency of ecosystem changes will increase in the next 25 to 50 years, and it is possible that changes will accelerate. Climate change can cause or exacerbate direct stress on ecosystems through high temperatures, a reduction in water availability, and altered frequency of extreme events and severe storms. Water temperatures in streams and rivers are likely to increase as the climate warms and are very likely to have both direct and indirect effects on aquatic ecosystems. Changes in temperature will be most evident during low flow periods when they are of greatest concern (NAST 2000). In some marine and freshwater systems, shifts in geographic ranges and changes in algal, plankton, and fish abundance are associated with high confidence with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation (IPCC 2007).

A warmer and drier climate is expected to result in reductions in stream flows and increases in water temperatures. Consequences could be a decrease in the amount of DO in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer-wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch et al. 2000). Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer et al. 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change.

While debated, researchers anticipate: 1) the frequency and intensity of droughts and floods will change across the nation; 2) a warming of about 0.4°F per decade; and 3) a rise in sea level

(NAST 2000). Sea level is expected to continue rising: during the twentieth century global sea level has increased 6 to 8 in.

As there is significant uncertainty in the rate and timing of change, as well as the effect of any changes that may be experienced in the action area due to climate change, it is difficult to predict the impact of these changes on sea turtles, Atlantic and Gulf sturgeon, smalltooth sawfish, and giant manta ray. The shrimp fisheries are expected to continue in the near and mid-term future in similar areas, at similar times, and with similar levels of effort, but there is no way to predict at this point in time whether the fishery resources and other environmental conditions will continue to support shrimp fisheries that are similar to the proposed action in the long-term future or indefinitely. Since the distribution of effort in the shrimp fisheries and the status of the resource can change over just a few years, we will primarily consider the effects of climate change on the listed species over the next 10 years. Longer-term effects of the fishery and climate change on ESA-listed species, whatever they may be, are speculative and difficult to extrapolate beyond 10 years.

Effects of Climate Change on Sea Turtles

Sea turtle species have persisted for millions of years. They are ectotherms, meaning that their body temperatures depends on ambient temperatures. Throughout this time they have experienced wide variations in global climate conditions and are thought to have previously adapted to these changes through changes in nesting phenology and behavior (Poloczanska et al. 2009). Given this, climate change at normal rates (i.e., thousands of years) is not thought to have historically been a problem for sea turtle species. At the current rate of global climate change, however, future effects to sea turtles are probable. Climate change has been identified as a threat to all species of sea turtles found in the action area (Conant et al. 2009; NMFS and USFWS 2013a; NMFS et al. 2011; Seminoff et al. 2015). Trying to assess the likely effects of climate change on sea turtles, however, is extremely difficult given the uncertainty in all climate change models, the difficulty in determining the likely rate of temperature increases, and the scope and scale of any accompanying habitat or behavior effects. In the Northwest Atlantic, specifically, loggerhead, green, and leatherback sea turtles are predicted to be among the more resilient species to climate change, while Kemp's ridley turtles are among the least resilient (Fuentes et al. 2013). Leatherbacks may be more resilient to climate change in the Northwest Atlantic because of their wide geographic distribution, low nest-site fidelity, and gigantothermy (Dutton et al. 1999; Fuentes et al. 2013; Robinson et al. 2009). Gigantothermy refers to the leatherbacks ability to use their large body size, peripheral tissues as insulation, and circulatory changes in thermoregulation (Paladino et al. 1990). Leatherbacks achieve and maintain substantial differentials between body and ambient temperatures through adaptations for heat production, including adjustments of the metabolic rate, and retention (Wallace and Jones 2008). However, modeling results show that global warming poses a "slight risk" to females nesting in French Guiana and Suriname relative to those in Gabon/Congo and West Papua, Indonesia (Dudley et al. 2016).

Sea turtles are most likely to be affected by climate change due to:

1. Changing air/land temperatures and rainfall at nesting beaches that could affect reproductive output including hatching success, hatchling emergence rate, and hatchling sex ratios;
2. Sea level rise, which could result in a reduction or shift in available nesting beach habitat, an increased risk of erosion and nest inundation, and reduced nest success;
3. Changes in the abundance and distribution of forage species, which could result in changes in the foraging behavior and distribution of sea turtle species as well as changes in sea turtle fitness and growth;
4. Changes in water temperature, which could possibly lead to a shift in their range, changes in phenology (timing of nesting seasons, timing of migrations) and different threat exposure; and
5. Increased frequency and severity of storm events, which could impact nests and nesting habitat, thus reducing nesting and hatching success.

Current approaches have limited power to predict the magnitude of future climate change, associated impacts, whether and to what extent some impacts will offset others, or the adaptive capacity of this species. Over the 10 years of the action addressed in this Opinion, sea surface temperatures are expected to rise less than 1°C. It is unknown if that is enough of a change to contribute to shifts in the range, distribution and recruitment of sea turtles or their prey. Theoretically, we expect that as waters in the action area warm, more sea turtles could be present or present for longer periods.

As climate continues to warm, feminization of sea turtle populations is a concern for many sea turtle species, which undergo temperature-dependent sex determinations. Rapidly increasing global temperatures may result in warmer incubation temperatures and higher female-biased sex ratios (Glen and Mrosovsky 2004; Hawkes et al. 2009). Increases in precipitation might cool beaches (Houghton et al. 2007), mitigating some impacts relative to increasing sand temperature. Though the predicted level of warming over the period of the action is small (i.e., <1°C), feminization occurs over a small temperature range (1-4°C) (Wibbels 2003) and several populations in the action area already are female biased (Gledhill 2007; Laloë et al. 2016; Patino-Martinez et al. 2012; Witt et al. 2010). The existing female bias among juvenile loggerhead sea turtles is estimated at approximately 3:2 females per males (Witt et al. 2010).

Feminization is a particular concern in tropical nesting areas where over 95% female-biased nests are already suspected for green turtles, and leatherbacks are expected to cross this threshold within a decade (Laloë et al. 2014; Laloë et al. 2016; Patino-Martinez et al. 2012). It is possible for populations to persist, and potentially increase with increased egg production, with strong female biases (Broderick et al. 2000; Coyne and Landry 2007; Godfrey et al. 1999; Hays et al. 2003), but population productivity could decline if access to males becomes scarce (Coyne 2000). Low numbers of males could also result in the loss of genetic diversity within a population. Behavioral changes could help mitigate the impacts of climate change, including shifting breeding season and location to avoid warmer temperatures. For example, the start of the nesting season for loggerheads has already shifted as the climate has warmed (Weishampel et al. 2004). Nesting selectivity could also help mitigate the impacts of climate on sex ratios as well (Kamel and Mrosovsky 2004).

At St. Eustatius in the Caribbean, there is an increasing female-biased sex ratio of green turtle hatchlings (Laloë et al. 2016). While this is partly attributable to imperfect egg hatchery practices, global climate change is also implicated as a likely cause as warmer sand temperatures at nesting beaches can result in the production of more female embryos. At this time, we do not know how much of this bias is also due to hatchery practices as opposed to temperature. Global warming may exacerbate this female skew. An increase in female bias is predicted in St. Eustatius, with only 2.4% male hatchlings expected to be produced by 2030 (Ibid). The study also evaluated leatherback sea turtles on St. Eustatius. The authors found that the model results project the entire feminization of the green and leatherback sea turtles due to increased air temperature within the next century (Ibid). The extent to which sea turtles may be able to cope with this change, by selecting cooler areas of the beach or shifting their nesting distribution to other beaches with smaller increases in sand temperature, is currently unknown.

Several leatherback nesting areas are already predominantly female, a trend that is expected to continue with some areas expecting at least 95% female nests by 2028 (Gledhill 2007; Laloë et al. 2016; Patino-Martinez et al. 2012). Hatchling success has declined in St. Croix (Garner et al. 2017), though there is some evidence that the overall trend is not climate or precipitation related (Rafferty et al. 2017). Excess precipitation is known to negatively impact hatchling success in wet areas, but can have a positive effect in dry climates (Santidrián Tomillo et al. 2015). In Grenada, increased rainfall (another effect of climate change) was found to have a cooling influence on leatherback nests, so that more male producing temperatures (less than 29.75°C) were found within the clutches (Houghton et al. 2007). There is also evidence for very wet conditions inundating nests or increasing fungal and mold growth, reducing hatching success (Patino-Martinez et al. 2014). Very dry conditions may also affect embryonic development and decrease hatchling output. Leatherbacks have a tendency towards individual nest placement preferences, with some clutches deposited in the cooler tide zone of beaches and have relatively weak nesting site fidelity; this may mitigate the effects of long-term changes in climate on sex ratios (Fuentes et al. 2013; Kamel and Mrosovsky 2004).

If nesting can shift over time or space towards cooler sand temperatures, these effects may be partially offset. A shift towards earlier onset of loggerhead nesting was associated with an average warming of 0.8°C in Florida (Weishampel et al. 2004). Early nesting could also help mitigate some effects of warming, but has also been linked to shorter nesting seasons in this population (Pike et al. 2006), which could have negative effects on hatchling output. Nesting beach characteristics, such as the amount of precipitation and degree of shading, can effectively cool nest temperatures (Lolavar and Wyneken 2015). However, current evidence suggests that the degree of cooling resulting from precipitation and/or shading effects is relatively small and therefore, even under these conditions, the production of predominantly female nests is still possible (Ibid). However, the impact of precipitation, as well as humidity and air temperature, on loggerhead nests is site specific and data suggest temperate sites may see improvements in hatchling success with predicted increases in precipitation and temperature (Montero et al. 2018; Montero et al. 2019). Conversely, tropical areas already produce 30% less output than temperate regions and reproductive output is expected to decline in these regions (Pike 2014).

Warming sea temperatures are likely to result in a shift in the seasonal distribution of sea turtles in the action area. In the northern part of the action area, sea turtles may be present earlier in the year if northward migrations from their southern overwintering grounds begin earlier in the spring. Likewise, if water temperatures are warmer in the fall, sea turtles could remain in the more northern areas later in the year. Potential effects of climate change include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson et al. 2009). McMahon and Hays (2006) reported that warming has caused a generally northerly migration of the 15°C sea surface temperature isotherm from 1983 to 2006. In response to this, leatherbacks have expanded their range in the Atlantic north by 330 km (Ibid). An increase in cold stunning of Kemp's ridley sea turtles in New England has also been linked to climate change and could pose an additional threat to population resilience (Griffin et al. 2019).

Furthermore, although nesting occurs in the south and mid-Atlantic (i.e., North Carolina and into Virginia), recent observations have caused some speculation that the nesting range of some sea turtle species may shift northward as the climate warms and that nest crowding may increase as sea level rises and available nesting habitat shrinks (Reece et al. 2013). Recent instances include a Kemp's ridley nesting in New York in July 2018 (96 hatchlings), a loggerhead nesting in Delaware in July 2018 (48 hatchlings), and a loggerhead nesting in Maryland in September 2017 (7 live hatchlings). The ability to shift nesting in time and space towards cooler areas could reduce some of the temperature-induced impacts of climate change (e.g., female biased sex ratio). Fuentes et al. (2020) modelled the geographic distribution of climatically suitable nesting habitat for sea turtles in the U.S. Atlantic under future climate scenarios, identified potential range shifts by 2050, determined sea-level rise impacts, and explored changes in exposure to coastal development as a result of range shifts. Overall, the researchers found that, with the exception of the northern nesting boundaries for loggerhead sea turtles, the nesting ranges were not predicted to change. Fuentes et al. (2020) noted that range shifts may be hindered by expanding development. They also found that loggerhead sea turtles would experience a decrease (10%) in suitable nesting habitat followed by green turtles. No significant changes was predicted in the distribution of climatically suitable nesting area for leatherbacks by 2050. Sea level rise is projected to inundate current habitats; however, new beaches will also be formed and suitable habitats could be gained, with leatherback sea turtles potentially experience the biggest gain in suitable habitat (Ibid).

Climate change may also increase hurricane activity, leading to an increase in debris in nearshore and offshore environments. This, in turn, could increase the occurrence of entanglements, ingestion of pollutants, or drowning. In addition, increased hurricane activity may damage nesting beaches or inundate nests with seawater. Increasing temperatures are expected to result in increased polar melting and changes in precipitation that may lead to rising sea levels (Titus and Narayanan 1995). Hurricanes and tropical storms occur frequently in the action area. They impact nesting beaches by increasing erosion and sand loss and depositing large amounts of debris on the beach. A lower level of leatherback nesting attempts occurred on sites more likely to be impacted by hurricanes (Dewald and Pike 2014). These storm events may ultimately affect the amount of suitable nesting beach habitat, potentially resulting in reduced productivity (TEWG 2007). These storms may also result in egg loss through nest destruction or inundation. Climate change may be increasing the frequency and patterns of hurricanes (IPCC 2014), which may result in more frequent impacts. These environmental/climatic changes could result in increased erosion rates along nesting beaches, increased inundation of nesting sites, a decrease in available nesting habitat, and an increase in nest crowding (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005; Reece et al. 2013). Changes in environmental and oceanographic conditions (e.g., increases in the frequency of storms, changes in prevailing currents), as a result of climate change, could accelerate the loss of sea turtle nesting habitat, and thus, loss of eggs (Antonelis et al. 2006; Baker et al. 2006; Conant et al. 2009; Ehrhart et al. 2014).

Tidal inundation and excess precipitation can contribute to reduce hatchling output, particularly in wetter climates (Pike 2014; Pike et al. 2015; Santidrián Tomillo et al. 2015). This is especially problematic in areas with storm events and in highly-developed areas where the beach has nowhere to migrate. Females may deposit eggs seaward of erosion control structures, potentially subjecting nests to repeated tidal inundation. A recent study by the USGS found that sea levels in a 620-mile "hot spot" along the East Coast are rising 3 to 4 times faster than the global average (Sallenger et al. 2012). In the next 100 years, the study predicted that sea levels will rise an additional 20-27 cm along the Atlantic coast "hot spot" (Ibid). The disproportionate sea level rise is due to the slowing of Atlantic currents caused by fresh water from the melting of the Greenland Ice Sheet. Sharp rises in sea levels from North Carolina to Massachusetts could threaten wetland and beach habitats, and negatively affect sea turtle nesting along the North Carolina coast. If warming temperatures moved favorable nesting sites northward, it is possible that rises in sea level could constrain the availability of nesting sites on existing beaches (Reece et al. 2013). There is limited evidence of a potential northward range shift of nesting loggerheads in Florida, and it is predicted that this shift, along with sea level rise, could result in more crowded nesting beaches (Ibid).

In the case of the Kemp's ridley, most of their critical nesting beaches are undeveloped and may still be available for nesting despite shifting landward. Unlike much of the Texas coast, the Padre Island National Seashore (PAIS) shoreline in Texas, where increasing numbers of Kemp's ridley are nesting, is accreting. Given the increase in nesting at the PAIS, as well as increasing and slightly cooler sand temperatures than at other primary nesting sites, PAIS could become an increasingly important source of males for a species, which already has one of the most restricted nesting ranges of all sea turtles. Nesting activity of Kemp's ridleys in Florida has also increased over the past decade, suggesting the population may have some behavioral flexibility to adapt to a changing climate (Pike 2013). Still, current models predict long-term reductions in sea turtle fertility as a result of climate change. These effects, however, may not be seen for 30-50 years because of the longevity of sea turtles (Davenport 1997; Hawkes et al. 2007; Hulin and Guillon 2007).

Changes in water temperature may also alter the forage base and, therefore, the foraging behavior of sea turtles (Conant et al. 2009). Likewise, if changes in water temperature affected the prey base for green, loggerhead, Kemp's ridley, or leatherback sea turtles, there may be changes in the abundance and distribution of these species in the action area. Depending on whether there was an increase or decrease in the forage base and/or a seasonal shift in water temperature, there could be an increase or decrease in the number of sea turtles in the action area. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as changes in salinity, light levels, and temperature (Duarte 2002; Saunders et al. 2013; Short and Neckles 1999). If seagrasses in the action area decline, it is reasonable to expect that the number of foraging green sea turtles would also decline as well. Rising water temperatures, and associated changes in marine physical oceanographic systems (e.g., salinity, oxygen levels, and circulation), may also impact the distribution/abundance of leatherback prey (i.e., jellyfish) and, in turn, impact the distribution and foraging behavior of leatherbacks (Attrill et al. 2007; Brodeur et al. 1999; NMFS and USFWS 2013; Purcell 2005; Richardson et al. 2009). Loggerhead sea turtles are thought to be generalists (NMFS and USFWS 2008), and, therefore, may be more resilient to changes in prey availability. As noted above, because we do not know the adaptive capacity of these individuals, or what level of temperature change would cause a shift in distribution, it is not possible to predict changes to the foraging behavior of sea turtles over the next 10 years. If sea turtle distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact to sea turtles due to the availability of food. Similarly, if sea turtles shifted to areas where different forage was available, and sea turtles were able to obtain sufficient nutrition from that new source of forage, any effect would be minimal. However, should climatic changes cause sea turtles to shift to an area or time where insufficient forage is available, impacts to these species would be greater. Despite site-specific vulnerabilities of the NWA DPS of loggerhead sea turtles, this DPS may be more resilient to changing climate than other management units (Fuentes et al. 2013). Van Houtan and Halley (2011) recently developed climate based models to investigate loggerhead nesting (considering juvenile recruitment and breeding remigration) in the Northwest Atlantic and North Pacific. These models found that climatic conditions and oceanographic influences explain loggerhead nesting variability. Specifically, the climate models alone explained an average 60% (range 18%-88%) of the observed nesting changes in the Northwest Atlantic and North Pacific over the past several decades. In terms of future nesting projections, modeled climate data predict a positive trend for Florida nesting (NWA DPS), with increases through 2040 as a result of the Atlantic Multidecadal Oscillation (Van Houtan and Halley 2011). In a separate model, Arendt *et al* (2013) suggested that the variability represents a lagged perturbation response to historical anthropogenic impacts. The nest count increases since 2008 may reflect a potential recovery response (Ibid).

Effects of Climate Change on Atlantic Sturgeon
Atlantic sturgeon have persisted for millions of years and have experienced wide variations in global climate conditions, to which they have successfully adapted. Climate change at historical rates (thousands of years) is not thought to have been a problem for sturgeon species. However, at the current rate of global climate change, future effects to Atlantic sturgeon are possible. Rising sea level may result in the salt wedge moving upstream in affected rivers. Atlantic sturgeon spawning occurs in fresh water reaches of rivers because early life stages have little to no tolerance for salinity. Similarly, juvenile Atlantic sturgeon have limited tolerance to salinity and remain in waters with little to no salinity. If the salt wedge moves further upstream, Atlantic sturgeon spawning and rearing habitat could be restricted. In river systems with dams or natural falls that are impassable by sturgeon, the extent that spawning or rearing may be shifted upstream to compensate for the shift in the movement of the salt wedge would be limited. While there is an indication that an increase in sea level rise would result in a shift in the location of the salt wedge, at this time there are no predictions on the timing or extent of any shifts that may occur; thus, it is not possible to predict any future loss in spawning or rearing habitat. However, in all river systems, spawning occurs miles upstream of the salt wedge. It is unlikely that shifts in the location of the salt wedge would eliminate freshwater spawning or rearing habitat. If habitat was severely restricted, productivity or survivability may decrease.

The increased rainfall predicted by some models in some areas may increase runoff and scour spawning areas and flooding events could cause temporary water quality issues. Rising temperatures predicted for all of the U.S. could exacerbate existing water quality problems with DO and temperature. While this occurs primarily in rivers in the southeast U.S. and the Chesapeake Bay, it may start to occur more commonly in the northern rivers. Atlantic sturgeon are tolerant to water temperatures up to approximately 82.4°F; these temperatures are experienced naturally in some areas of rivers during the summer months. If river temperatures rise and temperatures above this threshold are experienced in larger areas, sturgeon may be excluded from some habitats.

Increased droughts (and water withdrawal for human use) predicted by some models in some areas may cause loss of habitat including loss of access to spawning habitat. Drought conditions in the spring may also expose eggs and larvae in rearing habitats. If a river becomes too shallow or flows become intermittent, all Atlantic sturgeon life stages, including adults, may become susceptible to strandings or habitat restriction. Low flow and drought conditions are also expected to cause additional water quality issues. Any of the conditions associated with climate change are likely to disrupt river ecology causing shifts in community structure and the type and abundance of prey. Additionally, cues for spawning migration and spawning could occur earlier in the season causing a mismatch in prey that are currently available to developing sturgeon in rearing habitat.

Atlantic sturgeon in the action area are most likely to experience the effects of global climate change in warming water temperatures, which could change their range and migratory patterns. Warming temperatures predicted to occur over the next 100 years would likely result in a northward shift/extension of their range (i.e., into the St. Lawrence River, Canada) while truncating the southern distribution, thus affecting the recruitment and distribution of sturgeon rangewide. In the next 10 years, this increase in sea surface temperature is expected to be minimal, and thus, it is unlikely that any expanded or truncated range will be observed in the near future. If any shift does occur, it is likely to be minimal and thus, it seems unlikely that this small increase in temperature will cause a significant effect to Atlantic sturgeon or a significant modification to the number of sturgeon likely to be present in the action area over the life of the proposed action. However, even a small increase in temperate can affect DO concentrations. For example, a one degree change in temperature in Chesapeake Bay could make parts of Chesapeake Bay inaccessible to sturgeon due to decreased levels of DO (Batiuk et al. 2009).

Although the action area does not include spawning grounds for Atlantic sturgeon, sturgeon are migrating through the action area to reach their natal rivers to spawn. Elevated temperatures could modify cues for spawning migration, resulting in an earlier spawning season, and thus, altering the time of year sturgeon may or may not be present within the action area. This may cause an increase or decrease in the number of sturgeon present in the action area. However, because spawning is not triggered solely by water temperature, but also by day length (which would not be affected by climate change) and river flow (which could be affected by climate change), it is not possible to predict how any change in water temperature alone will affect the seasonal movements of sturgeon through the action area.

In addition, changes in water temperature may also alter the forage base and thus, foraging behavior of Atlantic sturgeon. Any forage species that are temperature-dependent may also shift in distribution as water temperatures warm and cause a shift in the distribution of Atlantic sturgeon. However, because we do not know the adaptive capacity of these species or how much of a change in temperature would be necessary to cause a shift in distribution, it is not possible to predict how these changes may affect foraging sturgeon. If sturgeon distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact on the availability of food. Similarly, if sturgeon shifted to areas where different forage was available and sturgeon were able to obtain sufficient nutrition from that new source of forage, any effect would be minimal. The greatest potential for effect to forage resources would be if sturgeon shifted to an area or time where insufficient forage was available; however, the likelihood of this happening seems low because sturgeon feed on a wide variety of species and in a wide variety of habitats.

Effects of Climate Change on Gulf Sturgeon

Climate change has potential implications for the status of the Gulf sturgeon through alteration of its habitat. Warmer water, sea level rise and higher salinity levels could lead to accelerated changes in habitats utilized by Gulf sturgeon. Saltwater intrusion into freshwater systems could negatively impact freshwater fish and wildlife habitat (FWC 2009) resulting in more saline inland waters that may eventually lead to major changes in inland water ecosystems and a reduction in the amount of available freshwater. Changes in water temperature may alter the growth and life history of fishes, and even moderate changes can make a difference in distribution and number (FWC 2009). Freshwater habitats can be stressed by changes in both water quality and levels because of anticipated extreme weather periods as mean precipitation is expected to decrease along with an increase in precipitation intensity. Both droughts and floods could become more frequent and more severe, which would affect river flow, water temperature, water quality, channel morphology, estuarine salinity regimes, and many other habitat features important to the conservation of Gulf sturgeon. Higher water temperatures combined with increased nutrients from storm runoff may also result in increased invasive submerged and emergent water plants and phytoplankton, which are the foundation of the food chain (FWC 2009). The rate that climate change and corollary impacts are occurring may outpace the ability of the Gulf sturgeon to adapt given its limited geographic distribution and low dispersal rate. As noted with aforementioned species, however, the expected small increase in temperature and its associated impacts over the next 10 years is unlikely to cause a significant effect to Gulf sturgeon.

Effects of Climate Change on Giant Manta Ray

Because the giant manta ray is 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). Climate change, however, may alter zooplankton abundance and distribution through the foreseeable future as a result of ocean acidification. As research to understand the exact impacts of climate change on marine phytoplankton and zooplankton communities is still ongoing, the severity of this threat to the giant manta ray has yet to be fully determined. Regardless, we have no information to indicate that the potential effects of climate change on giant manta ray will be anything but insignificant over the time frame of this Opinion (i.e., 10 years).

Effects of Climate Change on Smalltooth Sawfish

Sawfish are assumed to be at risk from climate change due to low intrinsic rates of population growth and slow rates of evolution (Field et al. 2009, Chin et al. 2010), although specific effects are hard to predict. Effects on sawfish habitat are clearer. Red mangroves and shallow (<1 m), euryhaline waters identified as habitat features essential for the conservation of smalltooth sawfish are likely to be affected by climate change, most notably through sea level rise, which increased by 0.19 m between 1901 and 2010 and is expected to increase 0.45 to 0.75 m by 2100 (IPCC 2013). Sea level increases would reduce the amount of shallow water available for juvenile smalltooth sawfish in areas where shorelines are armored (e.g., seawalls). Similarly, mangroves will be forced landward to remain at a preferred water inundation level and sediment surface elevation necessary for successful growth (Field 1995). Forced landward progression poses the greatest threat to mangroves in areas where there is limited or no room for landward or lateral migration due to shoreline armoring and coastal development (Semeniuk 1994). Reductions in the availability of shallow water or mangroves could have numerous ecological effects on sawfish, including increased sawfish predation, higher metabolic stress, and decreased body condition. Changes to air and water temperatures may affect both the species and the habitats it relies upon. Given that sawfish distribution is limited to areas with water temperatures above 8-12°C, warming could result in a northward range expansion for the species. Increased air temperature may also allow northward expansion of red mangroves, thus providing a primary habitat feature for the species outside of the current range. Warming may also affect precipitation patterns and tropical weather events. While increased rainfall could affect river discharges and salinity regimes and hurricanes could damage habitat features, the species has shown to be resilient to these types of stochastic events in the past. However, a change in the frequency or severity of these events could translate to additional effects on the species not previously considered or currently understood. In summary, we expect sea level rise resulting from climate change to be insignificant over the next 10 years, and unlikely to cause a significant effect to smalltooth sawfish, or its habitat, within this time frame.

5 EFFECTS OF THE ACTION

Effects of the action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (50 CFR 402.02).

In this section of our Opinion, we assess the effects of the continued action on listed species that are likely to be adversely affected. The analysis in this section forms the foundation for our jeopardy analysis found in Section 7 (Integration and Synthesis of Effects). 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 we are often forced to 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]). We generally select 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.

In this section, we assess the effects of the implementation of the sea turtle conservation regulations applicable to shrimp trawling and the authorization of southeast U.S. shrimp fisheries in federal waters on threatened and endangered species and designated critical habitat. Potential routes of effects of the proposed action on these species include fishing gear interactions and vessel interactions. Based on our understanding of the effects of the proposed action on these species, effects of the proposed action are expected to result only when listed species interact with active fishing gear. Smalltooth sawfish and sturgeon spend most of their time at or near the seafloor, where they are not subject to vessel interactions. Also, although sea turtles are susceptible to vessel strikes, shrimp trawl vessel strikes with sea turtles are extremely unlikely given the slow speed (2-3 knots) at which shrimp trawls are towed.

We quantify the effects to listed species in this section with estimates of bycatch/capture and mortality in an annual context to provide consistent metrics from a variety of available data sets. The time frame of this Opinion, however, is 10 years. Given the low observer effort in the shrimp fisheries, as well as further anticipated observer coverage lapses due to COVID-19 in the near future (i.e., 2020-2021), and the rarity of encounter for many of these species (e.g., Gulf sturgeon), we will later extrapolate out our take estimates over a 5-year time span. We do this to address high variability in bycatch rates combined with low observer effort potentially exacerbating annual estimates (i.e., "small numbers problem"). We discuss this further in Sections 7 (Integration and Synthesis of Effects) and 8 (ITS) of this Opinion.

5.1 Effects to Sea Turtles

Past Opinions on the shrimp fisheries, in particular the 2014 Opinion, provide extensive information on the types of interactions between sea turtles and shrimp trawls including: the effect of forced submergence on sea turtles unable to escape a shrimp trawl that lacks a TED or has an ineffective/illegal TED; sea turtle distribution, size/age class, and seasonality effects that may influence bycatch and bycatch mortality rates; the effect of repeat captures on sea turtles; and the reduction of sea turtle bycatch mortality due to the use of properly-installed and maintained TEDs. These discussions are incorporated herein by reference and summarized below.

General Effects of Forced Submergence

Shrimp trawling directly affects sea turtles, which are air-breathing reptiles. As turtles rest, forage, or swim on or near the bottom, shrimp trawls pulled at 1.5 to 3 knots can sweep over them. Shrimp trawls have an overhanging headrope to prevent shrimp from jumping over the mouth of the net when they are hit by the tickler chain or footrope. This overhang also hinders sea turtles from escaping shrimp trawls and heading for the surface. Video of wild loggerhead sea turtles encountering TEDs in trawls reveals that the sea turtles are usually oriented forward, apparently trying to outswim the advancing trawl footrope (NMFS 2002b and 2002c). Because of the trawl's greater speed or the sea turtles' eventually tiring, the sea turtles gradually fall back toward the rear of the net where they encounter a TED or, if a TED is not installed, where they fall into the cod end of the net and are caught. The vast majority of sea turtles that encounter trawls equipped with properly-functioning TEDs are able to escape quickly and can surface to breathe. Based on available information, actual physical contact with the trawl gear itself (e.g., netting, footrope, etc.) does not cause sea turtle injuries. Sea turtles encountering an improperly installed TED, however, may take longer to escape or be captured near or on the TED, depending on the extent of the issue. Upon retrieval of the trawl gear, captured sea turtles may be found dead, comatose, or alive, depending on time/extent of forced submergence effects. These effects include changes to blood chemistry and hormones resulting from stress, and metabolic acidosis caused by high lactic acid levels. Recovery can take several hours (Stabenau and Vietti 2003) to as many as 20 hours (Lutz and Dunbar-Cooper 1987), depending on the condition of the turtle (e.g., overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g., water temperature, wave action, etc.), and other factors. Stress effects from forced submergence may also negatively affect reproductive capability of female sea turtles.

In a worst-case scenario, sea turtles 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 drowning occurs, sea turtles may become comatose or unconscious, generally unresponsive, and with a drastically suppressed heart and respiration rate—indicative of at least a physiological injury. If resuscitated per the sea turtle conservation regulations (50 CFR 223.206(d)(1)(B)), some of these sea turtles may recover and survive. However, sea turtles caught in such condition and returned to the water without resuscitation are presumed to die (Kemmerer 1989).

Sea Turtle Distribution, Size/Age Class, and Seasonality Effects

The likelihood and frequency of sea turtle exposure to shrimp trawls is in large part a function of the extent of spatial and temporal overlap of each sea turtle species and fishing effort. Species' habitat preferences and the environmental conditions (i.e., water temperatures) may play a large part in the distribution and overlap of sea turtles and shrimp. In general, 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 likelihood and frequency that a sea turtle will be exposed to the gear. Size/age of sea turtles may result or vary the severity of capture and forced submergence effects; larger/older sea turtles have greater energy to try and escape from a trawl net, as well as greater respiratory capacity to withstand greater periods of forced submergence than smaller/younger sea turtles. Different species may react to forced submergence differently as well. Lastly, water temperature may result in differential effects of forced submergence, with a quicker escalation towards mortality in colder, winter temperatures (10-150 minutes) as compared to warmer, summer conditions (10-200 minutes) (Sasso and Epperly 2006)

Effects of Legal TED Use and TED Compliance on Sea Turtle Captures and Mortality

With its position just before the cod end in trawl nets, the use of TEDs in otter trawls has no effect on the likelihood or frequency of interactions, as TEDs do not serve as a physical or behavioral deterrent to sea turtles entering trawl nets. TEDs, however, do dramatically reduce the likelihood of interactions resulting in capture and bycatch mortality. Generally, sea turtles will orient/swim forward toward the trawl net opening when overtaken and continue swimming outward until overtaken and encountering the TED grid. After briefly exploring the area around the TED (usually searching upwards), sea turtles will find the escape opening and turn to exit the opening head-first. Based on our testing, sea turtle escape rates range from 30 to 118 seconds, depending on sea turtle size, TED design, and environmental conditions (NMFS 2002b and 2002c). As a result, we believe sea turtles escaping through properly functioning TEDs result in a very brief period of forced submergence that has very little physiological effect on sea turtles (Sasso and Epperly 2006; Stacy et al. 2016), including instances of repeat captures.[14]

Documented capture rates in early TED testing based on paired otter trawl tows (i.e., one with a TED and one without) conducted on chartered shrimp vessels documented a 97% exclusion efficiency rate (NMFS 1981). After further evaluation and testing of various TED designs, we determined that a perfectly installed and maintained TED will result in an approximately 95 to 98% turtle exclusion efficiency rate, depending on turtle size (J. Gearhart memorandum to S. Epperly, NMFS, March 29, 2011). The lower 95% efficiency rate was documented for smaller sea turtles used in our small sea turtle testing protocol between 2001-2010, which relied on 2- to 3-year-old juvenile turtles (26.5 to 39 cm SCL), while the higher 98% efficiency rate was documented in our wild turtle testing protocol between 2002-2007, which typically witnessed larger, adult turtles. Because the shrimp fisheries as a whole are prosecuted over a wide area and are more likely to interact with larger sea turtles on most shrimping grounds, we believe assuming a 97% exclusion efficiency rate as representative of the exclusion efficiency rate of compliant TEDs in the fleet (i.e., it is the mean of the two observed turtle exclusion efficiency rates) as we did in our past Opinions on the shrimp fisheries (e.g., NMFS 2002a; 2012a; 2014), is still appropriate.

The cited 97% exclusion efficiency rate is based on the shrimp fisheries complying with TED requirements, and this exclusion efficiency rate can be significantly reduced due to improper TED installation and use. In particular, TED grids installed at steep angles (i.e., angles greater than 55 degrees), can impede or prevent sea turtle release, resulting in drowning due to forced submergence. As such, we monitor TED compliance and have evaluated fisheries performance to ensure poor TED compliance does not impact our Opinion's conclusions. As cited in the 2014 Opinion, "future compliance levels are expected to result in TEDs being 88% effective, thus that level will be used as our compliance baseline." We have posted TED compliance information online, and the running database since 2014 can be found here: https://www.fisheries.noaa.gov/webdam/download/94029405. While TED compliance rates have varied historically and have been low at times, since 2016, the overall TED effective rate based on available data collected during TED inspections for any given month has not been lower than 90.83%. The overall TED effective rate for most months—including months with high numbers of inspections during the fishing season—has typically been around or above 95%. Based on the monthly TED inspection data referenced above, we believe poor TED compliance has not been an issue in the otter trawl shrimp fisheries in recent years.

While there is considerable uncertainty and potential bias (discussed in more detail below) with the compliance levels calculated based on TED inspection data, it is our primary source of compliance monitoring data from the fisheries and should continue to be used to support our

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[14] While this issue has been discussed in past Opinions on the shrimp fisheries as potentially being a more significant concern, we have reevaluated the available information, and no longer believe that multiple repeat captures in trawls with properly functioning TEDs is a noteworthy or quantifiable adverse effect on sea turtles. The issue is addressed in greater detail later in this Opinion.

analyses. Although the degree of uncertainty is unknown, and the inspection data may not provide an accurate or precise estimate of overall TED compliance in the fisheries, it should still provide an accurate measure of overall compliance trends in the fishery, which will provide valuable monitoring information that will help the agency determine if increased boardings are needed to improve overall compliance rates. Therefore, we will continue to monitor this metric and its potential influence on sea turtle bycatch mortality.

This Opinion does not rely on TED compliance as a surrogate metric for incidental take as was done in the 2014 Opinion. A past issue with calculating TED compliance is potential data bias when collected by law enforcement. This data is not strictly random in nature nor standardized by area, and law enforcement boardings may be targeted towards suspected offenders. This could skew the representative data downwards, and potentially present low TED compliance across the fisheries. It is also possible that the enforcement boarding data could present an overly optimistic picture of TED compliance, because as noted in prior shrimp opinions, TED compliance is likely lower when enforcement isn't looking. Instead of incorporating TED compliance in a surrogate, we use fishery observer data, which we consider as being random in nature due to the observer selection process. Consequently, we believe the information collected by fishery observers, particularly the condition of sea turtles incidentally captured that we then use to determine post-interaction mortality (PIM), better reflects actual fishery performance and sea turtle interactions without the aforementioned potential law enforcement related bias.

Effects of Post-Interaction Mortality

As documented in a July 7, 2020, memorandum (M. Barnette, NMFS SERO PRD Fishery Biologist, to D. Bernhart, NMFS SERO PRD Assistant Regional Administrator), the Southeast Sea Turtle Injury Workgroup reviewed all sea turtle interactions recorded by fisheries observers for the southeast U.S. shrimp trawl fisheries from 2012-2019 (n=232) to determine post-release injury and mortality percentages. The workgroup first determined if each interaction resulted from the current fishery interaction and then we followed Procedural Directive 02-110-21 "Process for Determining Post-Interaction Mortality of Sea Turtles Bycaught in Trawl, Net, and Pot/Trap Fisheries" to place the turtle into one of three mortality risk categories with associated post-release mortality rates, or provide justification for a 100% mortality determination (i.e., injuries or conditions that are incompatible with survival). Additional information on the development of the criteria is included in Stacy et al. (2016).

From the 2012-2019 observer records, the workgroup was unable to make determinations for 72 cases (19 loggerhead, 35 Kemp's ridley, 4 green, and 14 unknown or unidentified hardshell sea turtles) due to insufficient information. In many cases, observers may have recorded a sea turtle as captured and released alive, but there was no information on behavior, responsiveness, or activity level that would have allowed us to assign a condition category, as these interactions occurred prior to the training and requirement for observers to record these details. That is, for example, without any evaluative information following capture (e.g., activity level/responsiveness) or descriptive information on release (e.g., qualification of swimming/diving behavior) the workgroup was unable to determine if an animal should be scored with a Category I, Category II, or Category III condition. As a result, the workgroup classified these cases as unknown and excluded them from further analysis. For cases where observers documented comatose or unresponsive sea turtles upon capture, the workgroup used the capture condition as a proxy for final release condition. Thus, in cases where it was noted the resuscitation was performed or requested by the observer—regardless of the sea turtle's release condition—it was categorized as a mortality. The workgroup determined this was a conservative approach and reflective of normal fishery conditions (i.e., unknown whether the crew would initiate resuscitation on their own in the absence of guidance). As we intended to calculate bycatch mortality in the shrimp fisheries using a Bayesian model approach that utilizes observer data for direct mortality estimates (Babcock et al. 2018), the workgroup also excluded sea turtles that were determined to be dead upon capture and release to avoid double-counting and inflating PIM. That is, a turtle captured and released dead cannot possibly be subject to PIM. This resulted in 23 otter trawl records (8 Kemp's ridley, 12 green, 1 leatherback, and 1 unknown sea turtle) and 3 skimmer trawl records (1 Kemp's ridley and 2 green sea turtles) excluded from further consideration. Additionally, there were 4 records from 2012-2019 (1 loggerhead, 2 Kemp's ridley, and 1 green) that described moderately to severely decomposed animals not attributable to the observed interaction, which the workgroup omitted from further analysis. The workgroup ultimately made PIM determinations on 130 records from 2012-2019 for various trawl gears employed in the southeast U.S. shrimp fisheries. Specifically, they made PIM determinations for turtles captured by otter trawls (n=28), try nets (n=56), and skimmer and butterfly trawls (n=46). This information is summarized in Table 11, with more detailed information in Tables 12-16 below. Table 17 details the percentage of observer records where PIM evaluation was possible for each gear type.

TEDs are currently required in the observed otter trawl shrimp fisheries. Observers recorded sea turtles captured either in front of the TED (n=12), or behind the TED (n=46) in the cod end, as presented in Table 14. Individual sea turtles that were behind the TED had passed through the bars of the TED either due to their small body depth that allowed them to pass through the currently required 4-in TED bar spacing, or due to TED damage (bent or broken bars) that increased the TED bar spacing and allowed the turtle to pass through the TED. Of the 12 total cases of turtles captured in an otter trawl in front of the TED, the workgroup noted 3 instances (25%) that were due to what could be classified as gear issues. The first case was a leatherback sea turtle the observer reported was "too big to exit TED," the second case had a crab trap blocking the TED escape opening, and the third case was due to a twisted net that prevented the turtle from reaching the TED. While TEDs are not currently required in skimmer trawls, there was one small (22.5 cm CCL) green sea turtle captured by passing through the bars of the TED in a skimmer trawl that was operating under experimental fishing conditions with TEDs installed.

Table 11. The number of sea turtle observer records from 2012-2019 in each injury category by gear type, as well as the overall estimated PIM percentage by gear type. Calculations for estimating PIM are included below the category tallies for each gear type; standard rounding protocol is applied throughout this process.[15]

<table>
<tr><th></th><th>CATEGORY IA (10% MORTALITY)</th><th>CATEGORY IB (20% MORTALITY)</th><th>CATEGORY II (50% MORTALITY)</th><th>CATEGORY III (80% MORTALITY)</th><th>100% MORTALITY</th><th>TOTAL</th><th>ESTIMATED PIM (%)</th></tr>
<tr><td rowspan="2">OTTER TRAWL</td><td>11</td><td>0</td><td>1</td><td>7</td><td>9</td><td>28</td><td>58</td></tr>
<tr><td colspan="7">(11 x 0.10) + (0 x 0.20) + (1 x 0.50) + (7 x 0.80) + (9 x 1.0) = 16.2 / 28 = 0.5786 = 58%</td></tr>
<tr><td rowspan="2">TRY NET</td><td>43</td><td>6</td><td>3</td><td>2</td><td>2</td><td>56</td><td>19</td></tr>
<tr><td colspan="7">(43 x 0.10) + (6 x 0.20) + (3 x 0.50) + (2 x 0.80) + (2 x 1.0) = 10.6 / 56 = 0.1893 = 19%</td></tr>
<tr><td rowspan="2">SKIMMER/BUTTERFLY TRAWL</td><td>32</td><td>0</td><td>8</td><td>5</td><td>1</td><td>46</td><td>27</td></tr>
<tr><td colspan="7">(32 x 0.10) + (0 x 0.20) + (8 x 0.50) + (5 x 0.80) + (1 x 1.0) = 12.2 / 46 = 0.2652 = 27%</td></tr>
<tr><td rowspan="2">TOTAL</td><td>86</td><td>6</td><td>12</td><td>14</td><td>12</td><td>130</td><td>30</td></tr>
<tr><td colspan="7">(86 x 0.10) + (6 x 0.20) + (12 x 0.50) + (14 x 0.80) + (12 x 1.0) = 39 / 130 = 0.30 = 30%</td></tr>
</table>

Table 12. The number of sea turtle observer records from 2012-2019 in each injury category for all trawl gear captures combined, as well as the overall estimated PIM percentage by sea turtle species. Calculations for estimating PIM are included below the category tallies for each turtle species; standard rounding protocol is applied throughout this process.

<table>
<tr><th></th><th>CATEGORY IA (10% MORTALITY)</th><th>CATEGORY IB (20% MORTALITY)</th><th>CATEGORY II (50% MORTALITY)</th><th>CATEGORY III (80% MORTALITY)</th><th>100% MORTALITY</th><th>TOTAL</th><th>ESTIMATED PIM (%)</th></tr>
<tr><td rowspan="2">LOGGERHEAD</td><td>26</td><td>6</td><td>2</td><td>3</td><td>0</td><td>37</td><td>19</td></tr>
<tr><td colspan="7">(26 x 0.10) + (6 x 0.20) + (2 x 0.50) + (3 x 0.80) + (0 x 1.0) = 7.2 / 37 = 0.1946 = 19%</td></tr>
<tr><td rowspan="2">KEMP'S RIDLEY</td><td>41</td><td>0</td><td>10</td><td>10</td><td>8</td><td>69</td><td>36</td></tr>
<tr><td colspan="7">(41 x 0.10) + (0 x 0.20) + (10 x 0.50) + (10 x 0.80) + (8 x 1.0) = 25.1 / 69 = 0.3637 = 36%</td></tr>
<tr><td rowspan="2">GREEN</td><td>15</td><td>0</td><td>0</td><td>1</td><td>4</td><td>20</td><td>32</td></tr>
<tr><td colspan="7">(15 x 0.10) + (0 x 0.20) + (0 x 0.50) + (1 x 0.80) + (4 x 1.0) = 6.3 / 20 = 0.315 = 32%</td></tr>
<tr><td rowspan="2">UNIDENTIFIED</td><td>2</td><td>0</td><td>0</td><td>0</td><td>0</td><td>2</td><td>10</td></tr>
<tr><td colspan="7">(2 x 0.10) + (0 x 0.20) + (0 x 0.50) + (0 x 0.80) + (0 x 1.0) = 0.2 / 2 = 0.10 = 10%</td></tr>
<tr><td rowspan="2">LEATHERBACK</td><td>2</td><td>0</td><td>0</td><td>0</td><td>0</td><td>2</td><td>10</td></tr>
<tr><td colspan="7">(2 x 0.10) + (0 x 0.20) + (0 x 0.50) + (0 x 0.80) + (0 x 1.0) = 0.2 / 2 = 0.10 = 10%</td></tr>
<tr><td rowspan="2">TOTAL</td><td>86</td><td>6</td><td>12</td><td>14</td><td>12</td><td>130</td><td>30</td></tr>
<tr><td colspan="7">(86 x 0.10) + (6 x 0.20) + (12 x 0.50) + (14 x 0.80) + (12 x 1.0) = 39 / 130 = 0.30 = 30%</td></tr>
</table>

[15] While other Opinions have employed the approach of rounding up when dealing with any estimated fraction of an animal (i.e., you can't capture just a portion of an animal), given the basic modeling and various layers of extrapolations employed to estimate total fisheries bycatch in this Opinion, we employ a consistent, standardized rounding protocol (i.e., round half up) herein.

Table 13. The number of sea turtle observer records from 2012-2019 in each injury category for otter trawl captures, as well as the overall estimated PIM percentage by sea turtle species. Calculations for estimating PIM are included below the category tallies for each turtle species; standard rounding protocol is applied throughout this process.

	CATEGORY I (10% MORTALITY)	CATEGORY II (50% MORTALITY)	CATEGORY III (80% MORTALITY)	100% MORTALITY	TOTAL	ESTIMATED PIM (%)
LOGGERHEAD	1	0	0	0	1	10
	(1 x 0.10) + (0 x 0.50) + (0 x 0.80) + (0 x 1.0) = 0.10 / 1 = 0.10 = 10%					
KEMP'S RIDLEY	4	1	6	6	17	69
	(4 x 0.10) + (1 x 0.50) + (6 x 0.80) + (6 x 1.0) = 11.7 / 17 = 0.6882 = 69%					
GREEN	2	0	1	3	6	67
	(2 x 0.10) + (0 x 0.50) + (1 x 0.80) + (3 x 1.0) = 4 / 6 = 0.6666 = 67%					
LEATHERBACK	2	0	0	0	2	10
	(2 x 0.10) + (0 x 0.50) + (0 x 0.80) + (0 x 1.0) = 0.20 / 2 = 0.10 = 10%					
UNIDENTIFIED	2	0	0	0	2	10
	(2 x 0.10) + (0 x 0.50) + (0 x 0.80) + (0 x 1.0) = 0.20 / 2 = 0.10 = 10%					
TOTAL	11	1	7	9	28	58
	(11 x 0.10) + (1 x 0.50) + (7 x 0.80) + (9 x 1.0) = 16.2 / 28 = 0.5786 = 58%					

Table 14. The number of sea turtle observer records from 2012-2019 for otter trawl captures based on location in the net[1], as well as the overall estimated PIM percentage by sea turtle species.

	IN FRONT OF TED	BEHIND TED
LOGGERHEAD	1	0
KEMP'S RIDLEY	2	24
GREEN	0	19
LEATHERBACK	3	0
UNIDENTIFIED	6	3
TOTAL	12	46

[1] Numbers of turtles in this table include those where a PIM determination was not made (due to lack of evaluative information) but does not include those determined to be direct mortalities and excluded from further PIM consideration, hence the difference in totals compared to Tables 11-13.

The observer data regarding where sea turtles were captured in otter trawl nets reveal the significant number of Kemp's ridley and green sea turtles captured behind the TED were a result of the small size of the captured turtles. That is, the small body depth of captured juvenile Kemp's ridley and green sea turtles allow these turtles to pass through the standard 4-in bar spacing of TEDs utilized in the otter trawl fisheries and into the cod end of the net, versus passing out of the TED escape opening.

Table 15. The number of sea turtle observer records from 2012-2019 in each injury category for try net captures, as well as the overall estimated PIM percentage by sea turtle species. Only species observed as captures in try nets are included below. Calculations for estimating PIM are included below the category tallies for each turtle species; standard rounding protocol is applied throughout this process.

	CATEGORY IA (10% MORTALITY)	CATEGORY IB (20% MORTALITY)	CATEGORY II (50% MORTALITY)	CATEGORY III (80% MORTALITY)	100% MORTALITY	TOTAL	ESTIMATED PIM (%)
LOGGERHEAD	23	6	2	2	0	33	18
	(23 x 0.10) + (6 x 0.20) + (2 x 0.50) + (2 x 0.80) + (0 x 1.0) = 6.1 / 33 = 0.1848 = 18%						
KEMP'S RIDLEY	16	0	1	0	1	18	17
	(16 x 0.10) + (0 x 0.20) + (1 x 0.50) + (0 x 0.80) + (1 x 1.0) = 3.1 / 18 = 0.1722 = 17%						
GREEN	4	0	0	0	1	5	28
	(4 x 0.10) + (0 x 0.20) + (0 x 0.50) + (0 x 0.80) + (1 x 1.0) = 1.4 / 5 = 0.28 = 28%						
TOTAL	43	6	3	2	2	56	19
	(43 x 0.10) + (6 x 0.20) + (3 x 0.50) + (2 x 0.80) + (2 x 1.0) = 10.6 / 56 = 0.1893 = 19%						

Table 16. The number of sea turtle observer records from 2012-2019 in each injury category for skimmer (n=45) and butterfly (n=1) trawl captures, as well as the overall estimated PIM percentage by sea turtle species. Calculations for estimating PIM are included below the category tallies for each turtle species; standard rounding protocol is applied throughout this process.

	CATEGORY I (10% MORTALITY)	CATEGORY II (50% MORTALITY)	CATEGORY III (80% MORTALITY)	100% MORTALITY	TOTAL	ESTIMATED PIM (%)
LOGGERHEAD	2	0	1	0	3	33
	(2 x 0.10) + (0 x 0.50) + (1 x 0.80) + (0 x 1.0) = 1 / 3 = 0.3333 = 33%					
KEMP'S RIDLEY	21	8	4	1	34	30
	(21 x 0.10) + (8 x 0.50) + (4 x 0.80) + (1 x 1.0) = 10.3 / 34 = 0.3029 = 30%					
GREEN	9	0	0	0	9	10
	(9 x 0.10) + (0 x 0.50) + (0 x 0.80) + (0 x 1.0) = 0.9 / 9 = 0.10 = 10%					
TOTAL	32	8	5	1	46	27
	(32 x 0.10) + (8 x 0.50) + (5 x 0.80) + (1 x 1.0) = 12.2 / 46 = 0.2652 = 27%					

Table 17. The number of sea turtle observer records from 2012-2019 categorized as unknown compared to the total number of records, and the resulting percentage of records where evaluation was possible by gear type. Calculations are included below for each gear type; standard rounding protocol is applied throughout this process.

	UNKNOWN	TOTAL	EVALUATION POSSIBLE (%)
OTTER TRAWL	10	61	84
	61 – 10 = 51 / 61 = 0.8361 = 84%		
TRY NET	32	88	64
	88 – 32 = 56 / 88 = 0.6364 = 64%		
SKIMMER/BUTTERFLY TRAWL	30	79	62
	79 – 30 = 49 / 79 = 0.6203 = 62%		

TOTAL	72	228	68
	228 – 72 = 156 / 228 = 0.6842 = 68%		

Estimating the Extent of Effects: Otter Trawl Fisheries

Due to data limitations that have presented issues in calculating reasonable bycatch estimates noted in past Opinions (e.g., NMFS 2014), we further explored the ability to use observer data for calculating bycatch in the otter trawl fisheries. In our 2014 Opinion we concluded the bycatch estimates were "unacceptably uncertain to rely on them extensively in analyzing impacts, despite being based on the best available information." Since then, several studies have indicated that a Bayesian modeling approach can be effective at estimating bycatch in rate-event (i.e., data-limited) fisheries (Martin et al. 2015; Soldevilla et al. 2016). Thus, we employed available shrimp trawl fishery observer and effort data in a Bayesian modeling approach, which resulted in bycatch estimates as documented in Babcock et al. (2018), which we now use in estimating the total effect of the shrimp fisheries on sea turtle populations. The cumulative (2007-2015) bycatch and bycatch mortality (i.e., observed direct mortality) estimates for the Gulf of Mexico and South Atlantic calculated in Babcock et al. (2018) are presented in Tables 18 and 21 below.

Table 18. Gulf of Mexico otter trawl fisheries (try net and standard net) cumulative bycatch and observed bycatch mortality estimates 2007-2015 (Tables 14-15 in Babcock et al. 2018).

	BYCATCH		BYCATCH MORTALITY PRE-PIM APPLICATION	
	TRY NET	STANDARD NET	TRY NET	STANDARD NET
KEMP'S RIDLEY	2,037 (1,126-3,708)	3,939 (2165-7,685)	12 (0-69)	1,301 (646-2,687)
LOGGERHEAD	2,700 (1,722-4,212)	758 (299-1,542)	22 (1-118)	248 (96-556)
GREEN	532 (217-1,079)	1,275 (715-2,132)	4 (0-22)	418 (212-764)
UNKNOWN	687 (322-1,306)	1,500 (836-2,492)	5 (0-29)	443 (216-861)

Using the raw extrapolated bycatch estimates in Tables 18 and 21 for the Gulf of Mexico and South Atlantic, respectively, we then apply calculated PIM for each species as presented in Tables 15 and 13 above for try net and standard nets, respectively, to determine total bycatch mortality. In applying PIM, we first exclude the observed direct mortality calculated by Babcock et al. (2018) for each species to avoid double-counting. These calculations for each individual species are detailed in Table 19. For example, in the Gulf of Mexico region we subtract 12 Kemp's ridley sea turtles estimated as observed direct mortalities via try nets from the 2,037 total Kemp's ridley sea turtles estimated as try net bycatch, and apply 17% PIM as calculated in Table 15 above. This results in 344 Kemp's ridley sea turtles we estimate succumb to PIM following their capture and release from try nets. We add this back to the 12 observed direct mortalities to estimate a total of 356 Kemp's ridley sea turtle mortalities due to fisheries bycatch mortality in Gulf of Mexico try nets during the period 2007-2015. We followed the same approach for all sea turtle species. In Table 20 below, we calculate total annual bycatch mortality by dividing the totals for each species and gear in Table 19 by the number of years (i.e., 9 years for the Gulf of Mexico and 10 years for the South Atlantic) of observer data used in Babcock et al. (2018).

Finally, Table 24 below sums up the total annual estimated bycatch mortality in the southeastern U.S. otter trawl shrimp fisheries for each sea turtle species.

Table 19. Gulf of Mexico otter trawl fisheries total bycatch mortality estimates with applied PIM for 2007-2015. PIM application is based on 2012-2019 average PIM, all areas combined (Tables 15 and 13 above for try net and standard net, respectively). Standard rounding protocol is applied throughout this process.

	TRY NET	STANDARD NET
KEMP'S RIDLEY	2,037-12=2,025*0.17=344+12=356	3,939-1,301=2,638*0.69=1,820+1,301=3,121
LOGGERHEAD	2,700-22=2,678*0.18=482+22=504	758-248=510*0.10=51+248=299
GREEN	532-4=528*0.28=148+4=152	1,275-418=857*0.67=574+418=992
LEATHERBACK	-	-
UNKNOWN[1]	687-5=682*0.19=130+5=135	1,500-443=1,057*0.10=108+443=549

[1] For unknown species, the average PIM percentage across all species was utilized to estimate total mortality for try nets.

Table 20. Total annual estimated bycatch mortality for sea turtle species in the Gulf of Mexico otter trawl fisheries for 2007-2015.

	TRY NET	STANDARD NET	TOTAL
KEMP'S RIDLEY	356/9=40	3,121/9=347	387
LOGGERHEAD	504/9=56	299/9=33	89
GREEN	152/9=17	992/9=110	127
LEATHERBACK	-	-	-
UNKNOWN	135/9=15	549/9=61	76
TOTAL	**128**	**551**	**679**

Table 21. South Atlantic otter trawl fisheries (try net and standard net) cumulative bycatch and observed bycatch mortality estimates 2007-2015 (Tables 14-15 in Babcock et al. 2018).

	BYCATCH		BYCATCH MORTALITY PRE-PIM APPLICATION	
	TRY NET	STANDARD NET	TRY NET	STANDARD NET
KEMP'S RIDLEY	1,949 (811-4,212)	874 (177-3,085)	13 (1-91)	285 (55-1,059)
LOGGERHEAD	7,592 (4,468-12,099)	1,190 (315-3,897)	52 (2-292)	391 (99-1,330)
GREEN	424 (58-1,538)	551 (77-2,353)	3 (0-27)	179 (24-841)
UNKNOWN	-	1,501 (452-4,694)	-	495 (142-1,613)

Table 22. South Atlantic otter trawl fisheries total bycatch mortality estimates with applied PIM 2007-2015. PIM application is based on 2012-2019 average PIM, all areas combined (Tables 15 and 13 above for try net and standard net, respectively).

	TRY NET	STANDARD NET
KEMP'S RIDLEY	1,949-13=1,936*0.17=329+13=342	874-285=589*0.69=406+285=691
LOGGERHEAD	7,592-52=7,540*0.18=1,357+52=1,409	1,190-391=799*0.10=80+391=471
GREEN	424-3=421*0.28=118+3=121	551-179=372*0.67=249+179=428
UNKNOWN	-	1,501-495=1,006*0.10=101+495=596

Table 23. Total annual estimated bycatch mortality for sea turtle species in the South Atlantic otter trawl fisheries for 2007-2015.

	TRY NET	STANDARD NET	TOTAL
KEMP'S RIDLEY	342/10=34	691/10=69	103
LOGGERHEAD	1,409/10=141	471/10=47	188
GREEN	121/10=12	428/10=43	55
UNKNOWN	-	596/10=60	60
TOTAL	**187**	**219**	**406**

Table 24. Total annual estimated bycatch mortality for sea turtle species in the southeastern U.S. otter trawl fisheries for 2007-2015.

	GULF OF MEXICO		SOUTH ATLANTIC		TOTAL	
	TRY NET	STANDARD NET	TRY NET	STANDARD NET	TRY NET	STANDARD NET
KEMP'S RIDLEY	40	347	34	69	74	416
LOGGERHEAD	56	33	141	47	197	80
GREEN	17	110	12	43	29	153
UNKNOWN	15	61	-	60	15	121
TOTAL	**128**	**551**	**187**	**219**	**315**	**770**
TOTAL COMBINED TRY NET AND STANDARD NETS					**1,085**	

In summary, we anticipate the southeastern U.S. otter trawl shrimp fisheries on average currently results in 315 sea turtle mortalities (74 Kemp's ridley, 197 loggerhead, 29 green, and 15 unknown sea turtles) due to try net interactions and 770 sea turtle mortalities (416 Kemp's ridley, 80 loggerhead, 153 green, and 121 unknown sea turtles) due to standard net interactions, for a total annual average of 1,085 sea turtle mortalities due to fisheries bycatch. As mentioned above while discussing the results in Table 14, the high number of estimated mortalities for Kemp's ridley and green sea turtles (i.e., 416 and 153 sea turtles, respectively) in standard otter trawl nets is largely a function of the small size of encountered juvenile specimens of these 2 species, which allow them to pass through the standard 4-in bar spacing of TEDs currently required in the otter trawl fisheries. Additionally, the absence of leatherback and hawksbill sea turtle captures in the direct bycatch mortality estimates (Babcock et al. 2018) is a function of their general absence in the available observer data. We believe the shrimp fisheries may potentially encounter these sea turtles, though very rarely, and there is a likelihood of some level of bycatch, particularly over longer periods of time (e.g., 5 years). For the purposes of our take estimates and jeopardy analysis, we will assume some of the "unknown" sea turtle mortalities summarized in Table 24 will be leatherback or hawksbill sea turtles.

In order to estimate take of these 2 sea turtle species in the otter trawl fisheries, we will rely on sea turtle stranding data to provide insight into potential relative sea turtle distribution or species prevalence within southeast U.S. coastal areas where the shrimp fisheries are prosecuted. While the aforementioned "unknown" sea turtles noted by observers are likely unidentified hardshell sea turtles, we believe using sea turtle stranding data is the most reasonable approach for considering the potential rare bycatch of leatherback and hawksbill sea turtles in the shrimp fisheries. We complied STSSN data for each coastal state within the action area, excluding cold-stunned sea turtles and post-hatchlings, for the period of 2009-2019 in Table 25 below. The data indicates that of the approximately 40,000 stranded sea turtles identified by the STSSN, 0.70% we identified as leatherback and 0.92% were hawksbill sea turtles. There are obvious artifacts with relying on this data, particularly for leatherbacks, which have a more pelagic habitat preference and are typically not found close to shore off many states. Therefore, leatherbacks are less likely to strand before decomposition and other factors cause the carcass to sink before reaching the beach, as compared to other sea turtle species found in closer proximity to shore. Nevertheless, we again believe this stranding data represents the best available information and is an appropriate proxy to address the absence of leatherback and hawksbill sea turtles in the otter trawl observer data.

Table 25. Documented sea turtle strandings from the period 2009-2019 (cold-stuns and post-hatchlings excluded) for southeast U.S. states and species composition/representation for identified species (STSSN data via W. Teas, NMFS). Species in header: loggerhead (CC), green (CM), Kemp's ridley (LK), leatherback (DC), hawksbill (EI), and olive ridley (LO). Standard rounding protocol is applied throughout this process.

	CC	CM	LK	DC	EI	LO	UNIDENTIFIED	TOTAL	TOTAL IDENTIFIED
FLORIDA	8,219	10,550	2,264	100	244	2	324	21,703	21,379
ALABAMA	105	49	522	6	2	0	35	719	684
MISSISSIPPI	136	39	1,567	1	0	0	57	1,800	1,743
LOUISIANA	95	118	911	4	0	0	68	1,196	1,128
TEXAS	1,175	4,952	1,243	18	118	1	112	7,619	7,507
GEORGIA	925	310	362	28	1	0	6	1,632	1,626
SOUTH CAROLINA	951	207	294	71	0	0	18	1,541	1,523
NORTH CAROLINA	1,549	1,848	918	50	1	0	81	4,447	4,366
TOTAL	13,155	18,073	8,081	278	366	3	701	40,657	39,956
PERCENTAGE IDENTIFIED	32.92	45.23	20.22	0.70	0.92	0.01	-	-	100

Utilizing this stranding data, we adjust the combined bycatch (i.e., capture) numbers in Tables 18 and 21, and the bycatch mortality estimates in Table 24 to assign species identities to the unknown (i.e., unidentified to species) sea turtles in the observer data. For bycatch, we used the totals for unknown sea turtles in Table 18 for the Gulf of Mexico (687 for try net and 1,500 for standard net) and Table 21 for the South Atlantic (0 for try net and 1,501 for standard net) for a total of 687 try net captures and 3,001 standard net captures. We then used the percentages of identified stranded sea turtles in Table 25 to allocate captures and mortalities to specific sea turtle species. Of the 15 total annual unidentified sea turtles, we anticipate as bycatch mortality in the try net segment of the fishery yields an additional 5 loggerhead, 7 green, and 3 Kemp's ridley sea turtles. Likewise, of the 121 annual unidentified sea turtles, we estimate as bycatch mortality in the standard net segment of the fishery yields an additional 40 loggerhead, 55 green, 24 Kemp's ridley, 1 leatherback, and 1 hawksbill sea turtles. We applied a standard rounding protocol (i.e., round half up) throughout this process. These additional mortalities are added to the species-specific totals previously presented in Table 24 to calculate adjusted annual bycatch (captures) and bycatch mortality estimates in Table 26 below.

Table 26. Total adjusted annual estimated bycatch (captures) and bycatch mortality for sea turtle species in the southeastern U.S. otter trawl fisheries for 2007-2015; unknown captures and mortalities were allocated to species using STSSN data as discussed in the text. Standard rounding protocol is applied throughout this process.

	TRY NET		STANDARD NET	
	BYCATCH	MORTALITIES	BYCATCH	MORTALITIES
KEMP'S RIDLEY	4,212	77	5,801	440
LOGGERHEAD	10,603	202	3,305	120
GREEN	1,095	36	2,433	208
LEATHERBACK	5	0	21	1
HAWKSBILL	6	0	28	1
TOTAL	**15,921**	**315**	**11,588**	**770**

Effects of Ghost Captures and Repeated Captures

The 2014 Opinion included a discussion about "ghost captures" and turtles falling out of the trawl net during haulback thusly:

> Sea turtles that fail to escape through the TED can go undocumented by observers due to the animals falling out of non-compliant TEDs during haulback of the gear. This event is more likely to occur with high-angle TEDs (>55 degrees from the horizontal) than other types of violations because sea turtles can become impinged on deflector bars due to water pressure/flow against the carapace, particularly juveniles which have less strength to overcome drag. While "ghost captures" are less likely to occur with top-opening TEDs, SEFSC gear specialists have observed large-frame, top-opening TEDs without flotation rolling over (inverting) at the surface, which could also result in turtles falling out of the opening even in top-opening TEDs. In addition, some of the captured sea turtles may fall out of the front of the net as the lazy line is used to haul up the cod end of the net. These sea turtles may or may not be observed depending on conditions (e.g., high sea state or at night) and where the observer is positioned aboard the vessel. Waters fished for shrimp in the action area tend to be very murky, thus even turtles falling out near the surface can be easily missed.

In revisiting this issue, we anticipate there are potential scenarios where this could occur, such as: 1) if a small turtle got caught in the wing webbing; 2) if a TED was not installed properly or was illegally altered, such as a steep TED grid angle or if an escape opening was sewn shut; 3) if debris clogged a TED, preventing a turtle from escaping; or 4) if a "fresh" or late encounter turtle entered the trawl during haulback and hadn't made its way back to the TED yet. After examining these potential scenarios in more depth, we conclude based on available information that "ghost captures" occur so infrequently as not to result in any significant impact on affected sea turtle populations. While we have 2 instances of very small sea turtles becoming entangled in trawl netting, it was concluded this was due to improper flipper tagging protocols that resulted in the entanglement. We have already discussed TED compliance previously in this section, where available data indicate TED compliance levels are not significantly affecting our conclusions on TED performance. Excessive debris can present issues for sea turtle release, though available fishery observer data indicates this is a rare event; furthermore, any such instance has already been taken into consideration when conducting our take estimates and PIM evaluations since our current approach relies on observer data. Lastly, should a late-encounter turtle enter a trawl net and fall out during haulback, we believe that effect to be insignificant due to the very short duration of any such potential interaction (i.e., barring other factors, such as the aforementioned steep TED grids, etc.). In summary, upon reviewing available information and soliciting the professional opinion of our SEFSC gear experts, we conclude the aforementioned scenarios related to "ghost captures" occur so rarely as to prevent any measureable effect on sea turtle populations or simply result in insignificant effects on individual sea turtles.

Another potential consideration is repeated captures of sea turtles in shrimp trawls, should fishing effort be condensed into a discrete area where released turtles (via a TED) would be at risk of a second or third capture by nearby trawlers. As sea turtle populations recover and more sea turtles utilizing habitat where shrimp trawling activities occur, the total number of sea turtles exposed to incidental fishery bycatch increases, all other factors remaining the same. There is no available information, however, to indicate that the actual rate of recapture would be greater at higher sea turtle densities. As noted in Stacy et al. (2016), physiologic recovery from repeated (3 times) short duration forced submergence (7.5 minutes) in trawls equipped with functioning TEDs is possible with 3-hour intervals between interactions. Also, the number of times any individual sea turtle is caught is believed to be effort dependent (i.e., the more trawls fishing an area, the more times it is likely that an individual sea turtle will be captured). Thus, with the general declines in federally-permitted shrimp vessels, as well as a moratorium on the issuance of new limited-access federal shrimp permits in the Gulf of Mexico, and declines in shrimp fishery effort since 2003, the densities of trawlers on shrimp grounds have also likely declined. For these reasons, we believe the risk of repeated captures is probably much lower now than pre-2003. A significant unknown is a sea turtle's energy expenditure that might be associated with trying to outrun a trawl, before even encountering the TED. No data exist to quantify the extent of that effect. In summary, at this time we do not have sufficient information to calculate or quantify the potential impact of repeated captures that may be occurring, though we assume the impact of repeat captures is not a significant effect on individual sea turtles or sea turtle populations for the reasons discussed above.

Estimating the Extent of Effects: Non-Otter Trawl Fisheries

We believe the analysis included in the 2019 FEIS (NMFS 2019a) that analyzed alternatives to reduce the incidental bycatch and mortality of sea turtles in the southeast U.S. shrimp fisheries represents the best available information on the effects of skimmer trawls on sea turtle populations, herein incorporated by reference. The 2019 FEIS first extrapolated sea turtle take from 4 years of observer data (2012-2015) in the Gulf of Mexico, and synthesized available information from 2 studies for the North Carolina skimmer trawl fishery to estimate direct mortality. We then applied PIM to those estimates to obtain total bycatch mortality estimates for each area. The analysis in that document estimated 5,837 non-otter trawl vessels result in approximately 7,928 captures and 2,165 to 2,942 annual sea turtle mortalities due to fisheries bycatch. As a result, a final rule was published in December 2019 that would require all skimmer trawl vessels 40 ft and greater in length to use TEDs in their nets, effective April 1, 2021. As we previously noted, we delayed the effective date of this final rule until August 1, 2021, due to safety and travel restrictions related to the COVID-19 pandemic that prevented necessary training and outreach for fishers. We anticipate the rule will result in the conservation of 801-1,158 sea turtles annually. The FEIS anticipated the remainder of the non-otter trawl shrimp fisheries (i.e., vessels less than 40 ft in length that would still be operating under alternative tow time requirements) result in 1,364-1,784 sea turtle mortalities annually; the range is based on two different applications of PIM with different conservatism values (i.e., 27.2% and 36.8%) to calculate total mortalities. Since the release of that document, we refined the PIM analysis for non-otter trawl vessels, estimated to be 27% as presented in Table 11 above, which is essentially the same as the lower bounds used in the FEIS. For the take estimates in this Opinion, we will use the lower 27% PIM value in the FEIS, resulting in the anticipated bycatch mortality of 2,165 sea turtles prior to implementation of the TED requirements for skimmer trawls 40 ft and greater in length and 1,364 sea turtles after the implementation of the TED requirements for skimmer trawls 40 ft and greater in length.

The 2019 FEIS (NMFS 2019a) did not break down the estimated sea turtle mortalities by species. For the Gulf of Mexico, which represents the vast majority of skimmer trawl effort, observer data was collected on skimmer trawls operating from 2012-2015, during which 41 sea turtles were captured during a total of 2,699.23 observed hours. Two of these turtles were excluded, however, as their condition conclusively indicated they were previously dead and did not expire due to exposure in the observed skimmer trawl. Therefore, out of 39 captured turtles, 33 were Kemp's ridley (84.6%), 3 were green (7.7%), 2 were loggerhead (5.1%), and 1 was an unidentified presumed hard shell (2.6%) sea turtle. We will use this species distribution to estimate species-specific take for the combined Gulf of Mexico and South Atlantic (i.e., North Carolina) skimmer trawl fisheries, which results in 1,154 Kemp's ridley sea turtle mortalities (1,364 x 0.846 = 1,153.94) , 105 green sea turtles (1,364 x 0.077 = 105.03), 70 loggerhead sea turtle mortalities (1,364 x 0.051 = 6.96), and 35 unidentified sea turtles (1,364 x 0.026 = 35.46); we applied a standard rounding protocol (i.e., round half up) throughout this process. The number of estimated mortalities prior to implementation of the TED requirements for skimmer trawls 40 ft and greater in length and following the implementation of the TED requirement for skimmer trawl vessels 40 ft and greater in length are presented in Table 27 below. While the likelihood that the unidentified sea turtle was a Kemp's ridley sea turtle based on species abundance on the skimmer trawl fishing grounds, there is also a lesser possibility it could have been a green or loggerhead sea turtle. We do not expect that leatherback or hawksbill sea turtles interact with the skimmer trawl fisheries, because their preferred habitats do not overlap with the areas where skimmer trawls operate. We also do not believe the stranding data presented in Table 25 for the otter trawl fisheries is an appropriate proxy for the skimmer trawl fisheries because it lacks accurate stranding data from Louisiana due to geographical constraints (i.e., remote and numerous marshy bayous and bays that prevent regular surveys), and this area is where the majority of skimmer trawl effort occurs. Therefore, we will allocate the 35 unidentified (presumed hard shell) sea turtles amongst the 3 observed and identified sea turtle species (38 total turtles identified by species) captured in the skimmer trawl fisheries (i.e., Kemp's ridley, green, and loggerhead sea turtles in the proportions they have been observed captured in skimmer trawls). Isolating the unidentified sea turtles, yields a species representation of 86.8% for Kemp's ridley sea turtles (33 / 38 = 0.8684), 7.9% for green sea turtles (3 / 38 = 0.0789), and 5.3% for loggerhead sea turtles (2 / 38 = 0.0526). Allocating the 35 unidentified sea turtles amongst these three species at the respective percentages yields 30 Kemp's ridley sea turtles (35 x 0.868 = 30.38), 3 green sea turtles (35 x 0.079 = 2.77), and 2 loggerhead sea turtles (35 x 0.053 = 1.86); we applied a standard rounding protocol (i.e., round half up) throughout this process. The adjusted mortalities post-rule are represented in the last column of Table 27.

Table 27. Total annual estimated bycatch mortality for sea turtle species in the southeastern U.S. skimmer trawl fisheries.

	MORTALITIES PRE-RULE	MORTALITIES POST-RULE	ADJUSTED MORTALITIES POST-RULE
KEMP'S RIDLEY	1,832	1,154	1,184
GREEN	167	105	108
LOGGERHEAD	110	70	72
UNIDENTIFIED	56	35	-
TOTAL	**2,165**	**1,364**	**1,364**

Effects of Legal TED Exemptions

Trawls exempted from both TED use and tow times are limited to roller-frame trawls. It is unlikely that a sea turtle would become entrapped within a roller-frame trawl due to the required deflector bars positioned across the trawl mouth (Epperly et al. 2002), thus this exemption is not expected to have any adverse results on sea turtles. Tow-time requirements for vessels currently exempted from TED use (including permitted vessels conducting scientific research) are expected to reduce effects to the extent that they are complied with. We expect shrimp trawlers conducting scientific research comply with the tow time requirements as they typically have short tow times (e.g., 30 minutes) and they need to be consistent to ensure standardized data. The sea turtle conservation regulations specify that for those limited circumstances where shrimpers may comply with tow time limits instead of using TEDs, tow times be limited to 55 minutes from April through October and to 75 minutes from November through March (50 CFR 223.206(d)((3)). These regulations were based on the National Research Council (NRC) findings that sea turtle death rates in trawls are near zero until tow times exceed 60 minutes (NRC 1990). Tow time is measured from the time that the trawl door enters the water until it is removed from the water. For a trawl that is not attached to a door, the tow time is measured from the time the cod end enters the water until it is removed from the water.[16] The regulatory tow time limits include a 15-minute allowance for setting and retrieving gear, since the NRC analysis of tow times looked at bottom time only. Thus, the summer and the winter regulatory

[16] With the pending effectiveness of the December 20, 2019, final rule (84 FR 70048) requiring skimmer trawls 40 ft and greater in length to use TEDs, this definition will change to read "For a trawl that is not attached to a door, the tow time begins at the time the codend enters the water and ends at the time the codend is emptied of catch on deck."

limits, with adequate compliance, are expected to result in near zero sea turtle deaths because of expected bottom times under 60 minutes.

Effects of Other Sea Turtle Conservation Regulations
The sea turtle conservation regulations also require fishers to attempt to resuscitate comatose sea turtles (50 CFR 223.206(d)(1)(B)) before returning them to the water. Fishing in compliance with the sea turtle conservation regulations since 2002 likely has resulted in fewer sea turtles caught in need of resuscitation. In cases where sea turtles are comatose from capture, these regulations allow for some of these turtles to recover and be released alive with increased chances of survival. It is unclear to what extent shrimp fishers comply with the resuscitation requirements. Despite our outreach efforts, as well as that of our state partners, anecdotal reports suggest that many fishers return captured sea turtles to the water immediately because they fear there may be consequences of having a listed sea turtle on deck if boarded by law enforcement authorities. As a result, this Opinion explores additional strategies to improve proper sea turtle handling and resuscitation in the shrimp fisheries (see Section 9).

5.2 Effects to Atlantic Sturgeon

Effects of shrimp trawling on Atlantic sturgeon are expected to result from physical interactions with otter trawl gear use in the South Atlantic federal shrimp fishery. The otter trawl is the only gear type used to harvest shrimp species in South Atlantic federal waters. Otter trawls are classified as active fishing gear because animals do not voluntarily enter the gear; they are either swept up from the seabed or netted from the water by the gear (NRC 2002). In this manner, Atlantic sturgeon that are foraging within or moving through an active trawling location may be captured via envelopment or entanglement in the trawl's netting and subsequently injured or killed. Atlantic sturgeon may also escape through TEDs unobserved. While this could greatly increase the survival of Atlantic sturgeon interacting with shrimp trawls, it could also result in stress or injury to individuals escaping through the TED.

The ASMFC (2007) reported on Atlantic sturgeon bycatch in various types of fishing gear. They determined that there are no significant differences in bycatch in otter trawls based on the mesh size classes that were observed, although meshes in the range of 100-150 mm may be moderately more likely to be associated with Atlantic sturgeon bycatch. The ASMFC found the greatest correlation between Atlantic sturgeon bycatch and depth fished with otter trawls. The majority (84%) of Atlantic sturgeon bycatch in otter trawls occurred at depths less than 20 m, and about 90% of bycatch was observed at depths less than 30 m.

Because different life stages of Atlantic sturgeon are associated with different habitat types and water depths, the likelihood and frequency of Atlantic sturgeon interactions varies by life stage. Only trawl interactions with adult and sub-adult Atlantic sturgeon are expected because younger life stages do not enter the marine environment. Adult Atlantic sturgeon will reside in the marine habitat during the non-spawning season and forage extensively. Coastal migrations by adult Atlantic sturgeon are extensive and are known to occur over sand and gravel substrate

(Greene et al. 2009). Atlantic sturgeon remain in the marine habitat until the waters begin to warm, at which time ripening adults migrate back to their natal rivers to spawn. Sub-adult Atlantic sturgeon also utilize the marine environment for foraging and for migration between estuaries and bays. Trawl surveys conducted off Virginia and North Carolina between 1988 and 2006 as part of the Cooperative Winter Tagging Cruises captured primarily sub-adult Atlantic sturgeon (141 sub-adults out of 146 total Atlantic sturgeon captures) (Laney et al. 2007). Laney et al. (2007) reported that this could either be due to the age structure of the Atlantic sturgeon population or to gear selectivity, with adult Atlantic sturgeon better able to swim away and escape capture.

Estimating the Extent of Effects[17]

We have received reports from the mandatory federal observer program of 13 Atlantic sturgeon captures in the South Atlantic shrimp trawl fisheries from 2008-2020 (Table 28); 7 of these sturgeon were captured by a single shrimp trawler off Winyah Bay, South Carolina, from October 27-29, 2008 (E. Scott-Denton, NMFS, pers. comm.). All captures occurred within state waters in 14-38 ft of water (Figure 8). The majority of observed captures occurred in standard trawl nets, while 2 were observed in try nets. Of the documented standard net captures, 27% were recorded as dead (n=3) while 73% were released alive; overall, available information indicates 77% of all captured Atlantic sturgeon in the South Atlantic shrimp fisheries were released alive. We have no available information on PIM for Atlantic sturgeon, but PIM risks and consequences for fish species are not the same as they are for sea turtles. While delayed mortality of Atlantic sturgeon is conceivable from injuries sustained by interactions with shrimp trawls, the most significant effect pathway resulting in PIM for sea turtles is forced submergence. Because Atlantic sturgeon do not breathe air, forced submergence in shrimp trawls does not present the same concern.

Table 28. Documented bycatch of Atlantic sturgeon in the shrimp fisheries based on NMFS Observer Program data (E. Scott-Denton, NMFS, pers. comm.).

YEAR	LOCATION	DEPTH (FT)	NET	STATUS
2008	SC STATE WATERS	18.2	STANDARD NET	ALIVE
2008	SC STATE WATERS	20.7	STANDARD NET	ALIVE
2008	SC STATE WATERS	18	TRY NET	ALIVE
2008	SC STATE WATERS	25.3	STANDARD NET	ALIVE
2008	SC STATE WATERS	25.3	STANDARD NET	DEAD
2008	SC STATE WATERS	25.3	STANDARD NET	ALIVE
2008	SC STATE WATERS	29.3	STANDARD NET	ALIVE
2011	SC STATE WATERS	35	STANDARD NET	ALIVE
2011	GA STATE WATERS	17	STANDARD NET	ALIVE
2011	GA STATE WATERS	14	STANDARD NET	ALIVE
2016	NC STATE WATERS	38	STANDARD NET	DEAD
2016	NC STATE WATERS	27	STANDARD NET	DEAD
2020	SC STATE WATERS	N/A	TRY NET	ALIVE

[17] We only estimate the effect of the fisheries in federal waters, where we permit vessels and authorize shrimp fishing activity.

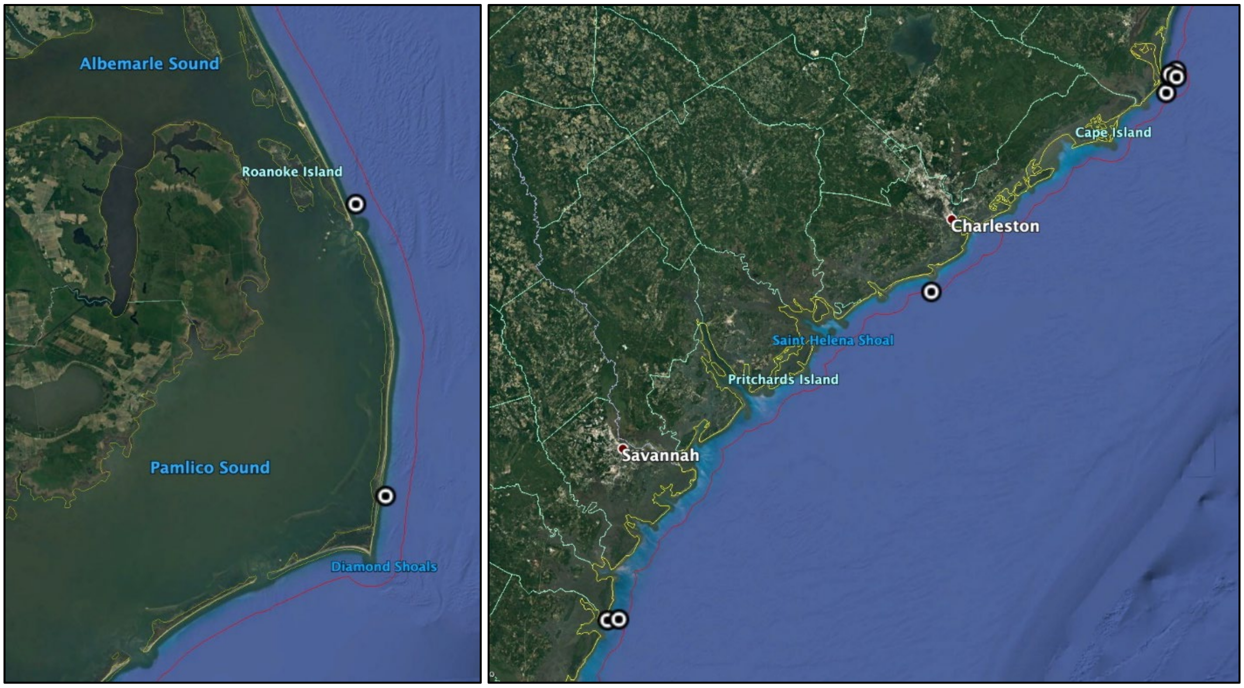

Figure 8. Locations of Atlantic sturgeon captures in the shrimp fisheries based on NMFS Observer Program data (E. Scott-Denton, NMFS, pers. comm.).

The federal fishery observer program became mandatory in the South Atlantic federal shrimp fishery in 2008. While observers in the program are only required on federally-permitted vessels, most vessels are known to fish both state and federal waters on any given trip. Therefore, fishery observers on federally-permitted vessels also observe fishery effort in state waters. The new mandatory program made observer data more random and unbiased, and better suited for use in fishery statistics. For 2008 to 2019, South Atlantic shrimp fisheries were observed on 592 trips over 1,803 sea days (E. Scott-Denton, NMFS, pers. comm.). From 2008 to 2019, observers documented 11 Atlantic sturgeon captures in standard otter trawl nets by South Atlantic shrimp trawlers during the 592 observed trips, for a bycatch rate of 11 sturgeon/592 trips or 0.0186 Atlantic sturgeon per trip. Available effort information indicates South Atlantic shrimp fisheries averaged 14,417 trips per year during the 2014-2018 period (M. Travis, NMFS, pers. comm. via ACCSP). Multiplying the fishing effort by the bycatch rate (14,417 trips x 0.0186 sturgeon/trip), we estimate that 268 Atlantic sturgeon were captured in otter trawl gear by shrimp trawls per year during the 2014-2018 period.

Atlantic sturgeon can also be captured in try net gear used in shrimp fisheries. From 2008-2020, 2 Atlantic sturgeon were captured in try nets by South Atlantic shrimp trawlers in state waters during the 592 observed trips, for a bycatch rate of 2 sturgeon/592 trips or 0.0034 Atlantic sturgeon/trip. Extrapolating take out as we did for standard nets, we estimate 49 Atlantic sturgeon (14,417 trips x 0.0034 sturgeon/trip) were captured annually in try nets by South Atlantic shrimp fisheries during the 2014-2018 period. This estimate, combined with the 268 estimated standard net captures, yields a total estimate of 317 annual Atlantic sturgeon captures in the South Atlantic shrimp fisheries in recent years.

All of the reported captures of Atlantic sturgeon in the South Atlantic shrimp trawls occurred in state waters; however, it is not uncommon for fishers to trawl between state and federal waters. The vast majority of observed shrimping in the South Atlantic has occurred in state waters, and this is representative of the entire South Atlantic shrimp fleet. From June 2008 through November 2019, there have been 3,676 observed tows for the South Atlantic shrimp fisheries (E. Scott-Denton, NMFS, pers. comm.). Of these observed tows, 3,139 (85.4%) occurred in state waters, while only 453 (12.3%) were observed in the EEZ; 84 tows (2.3%) recorded no latitude/longitude position information. It is likely that the catch rates of sturgeon are higher in state waters than federal waters. This is based on the species' apparent preference for coastal, shallower waters, and the actual observed catches—13 in state waters versus 0 in federal waters, even with more sampling occurring in state waters than federal waters. We do know, however, that Atlantic sturgeon are caught by other trawl fisheries in the EEZ, and we believe that the federal shrimp fishery catches them too. Therefore, using a combined state-federal CPUE is reasonable and conservative, and we will assume that the percentage of the total estimated sturgeon captures in federal waters is the same as the percentage of effort observed in state waters. We estimate that about 12.3% of the 317 estimated captures of Atlantic sturgeon by otter trawl gear, or 39 (6 try net captures [49 x 0.123 = 6.027] and 33 standard net captures [268 x 0.123 = 32.964]) estimated captures of Atlantic sturgeon by South Atlantic shrimp fisheries in otter trawl gear, may have occurred in federal waters.

The total number of sturgeon that interact with otter trawl gear in the shrimp fisheries is likely much higher than simply the number of Atlantic sturgeon observed captured in shrimp trawls, as anecdotal reports and scientific research indicate that Atlantic sturgeon may escape through TEDs. Flexible flatbar flynet TED testing was conducted in North Carolina from 2008-2009 by our SEFSC Pascagoula Laboratory to evaluate catch loss aboard contracted commercial vessels utilizing the trouser trawl technique (NMFS 2012c). A standard 85-ft flynet trawl was modified to accommodate 2 separate cod ends with a divider panel originating at the cod end split and extending into the body of the trawl; the Pascagoula Laboratory opted for this technique because of the high between-tow catch variability associated with flynet trawls. The TED was installed in one cod end, while no TED was installed in the other net to serve as a control. Video obtained from a camera mounted behind the TED opening revealed several Atlantic sturgeon escaping through the TED opening. In the course of 4 tows, the control net (with no TED) captured a total of 15 sturgeon, while the net with the TED captured only 2 Atlantic sturgeon. Based on this data, the TED resulted in an 87% reduction in Atlantic sturgeon bycatch by number of individuals (i.e., 2 Atlantic sturgeon were captured and 13 were assumed to have escaped capture through the TED out of an estimated 15 Atlantic sturgeon encountering the trawl gear; i.e., 13% capture rate). The documented 95% reduction by weight of Atlantic sturgeon during the study also suggests that captured Atlantic sturgeon are smaller individuals. We applied this information to our previous estimate of 33 incidental captures of Atlantic sturgeon by standard nets in federal waters; try nets do not have TEDs and all captures are observed). We believe this estimate is likely only 13% of the total number of Atlantic sturgeon interacting with the federal fishery. We, therefore, estimate that a total of 254 Atlantic sturgeon (i.e., 33 captured ÷ 13% of the total interactions) interacted with the South Atlantic federal shrimp fishery based on the 2009 effort data, with 13% (33 Atlantic sturgeon) incidentally captured in shrimp nets and 87% (221 Atlantic sturgeon) escaping through TEDs unobserved.

Studies in a variety of fisheries have shown that mortality of Atlantic sturgeon incidentally caught in trawl gear is very low, with most surveys showing 0% mortality (e.g., Stein et al. 2004). Based on observer data from South Atlantic shrimp fisheries, 3 mortalities were observed out of the 11 Atlantic sturgeon incidentally captured by standard otter trawl nets between 2008 and 2020 (Table 28), for a calculated mortality rate of 27%. This rate is high compared to most reports for trawl fisheries. It may be an artifact of the low number of observed incidental captures of Atlantic sturgeon in shrimp trawl fisheries, or it may reflect some difference between shrimp trawling and other trawl fisheries, perhaps an effect of warmer, southern waters. In any event, using this apparently high mortality rate will be a conservative approach. Applying the estimated mortality rate to the estimated annual incidental captures of Atlantic sturgeon in federal waters between 2014-2018, we estimate that 27% of the 33 Atlantic sturgeon, or 9 Atlantic sturgeon, incidentally captured in federal waters would die after their capture. There was no observed mortality for sturgeon captured in try nets, most likely due to the fact that try nets are generally pulled for short periods to determine the fishability and productivity of an area. Based on the short tow times and lack of observed mortality, we do not believe Atlantic sturgeon captured in try nets will be killed. We have no information to evaluate or estimate PIM issues in Atlantic sturgeon, but as already discussed, it is not thought to present an important concern for the species.

Next, we need to evaluate the effects on the 5 Atlantic sturgeon DPSs using MSA percentages of each DPS in the southeast we presented in Table 6 (Section 3.2.7). We offer the results in Table 29 below. Lacking any other information, we assume the percentages of each DPS in federal waters are representative of the percentages of each DPS actually captured in the fisheries.

Table 29. Total annual estimated interactions, bycatch, and mortalities for Atlantic sturgeon in the South Atlantic federal shrimp fishery. Standard rounding protocol is applied throughout this process.

DPS (MSA %)	Total Interactions	Try Net Bycatch/Mortalities	Standard Net Bycatch/Mortalities	Total Bycatch/Mortalities
Gulf of Maine DPS (1.0%)	3 (300 x 0.01)	0/0 (6 x 0.01)	0/0 (33/9 x 0.01)	0/0 (39/9 x 0.01)
New York Bight DPS (3.6%)	11 (300 x 0.036)	0/0 (6 x 0.036)	1/0 (33/9 x 0.036)	1/0 (39/9 x 0.036)
Chesapeake Bay DPS (9.6%)	29 (300 x 0.096)	1/0 (6 x 0.096)	3/1 (33/9 x 0.096)	4/1 (39/9 x 0.096)
Carolina DPS (33.8%)	101 (300 x 0.338)	2/0 (6 x 0.338)	11/3 (33/9 x 0.338)	13/3 (39/9 x 0.338)
SA DPS (52.9%)	159 (300 x 0.529)	3/0 (6 x 0.529)	17/5 (33/9 x 0.529)	21/5 (39/9 x 0.529)
Atlantic Sturgeon Total[1]	300	6/0	33/9	39/9

[1] Note that the total bycatch and mortality of each category by DPS may be different than bycatch and mortality of Atlantic sturgeon as a whole due to rounding issues.

In summary, we estimate the South Atlantic federal shrimp fishery results in 300 total interactions with Atlantic sturgeon that corresponds to 39 bycatch captures and 9 mortalities, annually, based on available data.

5.3 Effects to Gulf Sturgeon

In general, we expect any effects to Gulf sturgeon resulting from interactions with shrimp trawls to be similar to that as discussed for Atlantic sturgeon in Section 5.2. Based on our knowledge of Gulf sturgeon and shrimp trawling in the Gulf of Mexico, however, the temporal and spatial overlap of Gulf sturgeon and the federal Gulf of Mexico shrimp fishery is limited. Only adult Gulf sturgeon migrate into marine waters; other life stages have little to no movement into marine waters. Adult Gulf sturgeon are only susceptible to interaction with shrimp trawls during November through February, when they are feeding in the northern Gulf of Mexico and the area known as the “Big Bend” off Florida. During those winter months, because Gulf sturgeon are demersal, they are likely to be captured by shrimp trawls in those areas that drag their nets along the seafloor. Data describing the Gulf sturgeon’s swimming ability in the Suwannee River strongly indicated that they cannot continually swim against prevailing currents of greater than 1 to 2 m per second (Wakeford 2001). Thus, even though shrimp trawls travel through the water at slow speeds, it is still highly unlikely that all Gulf sturgeon would be able to outswim a shrimp trawl. Relocation data indicate most Gulf sturgeon prefer sandy shoreline habitats in more shallow waters. The depth of the tow resulting in the single observed Gulf sturgeon capture in federal waters was much deeper (56.8 ft) than where Gulf sturgeon have been previously documented, showing that interactions can occur in waters deeper than previously believed. Such deep water interactions are still thought to be very rare, however, and the best available data indicate most Gulf sturgeon remain inshore of where the federal fishery is prosecuted.

Based on the available information, as with Atlantic sturgeon, TED requirements in shrimp otter trawl fisheries likely benefit Gulf sturgeon by providing a route of escape. Results of the flexible flatbar flynet TED testing for Atlantic sturgeon indicated their bycatch was reduced by 87%. Therefore, the mandatory use of TEDs in the Gulf of Mexico shrimp otter trawl fisheries likely decreases the number of Gulf sturgeon captured in shrimp trawls significantly, and most Gulf sturgeon encountering shrimp trawls in both state and federal shrimp fisheries will escape the nets alive.

Estimating the Extent of Effects[18]

The federal fishery observer program was voluntary between 1992 through June 2007, with coverage typically less than 1% of total shrimp effort. No Gulf sturgeon captures were observed during that period. Mandatory observer coverage was initiated in the Gulf of Mexico shrimp fishery in July 2007 and since then only 2 Gulf sturgeon have been observed captured, one in federal waters and one in state waters. Both of these captures were in standard trawl nets in relatively shallow waters.

We have other information indicating Gulf sturgeon are vulnerable to capture in trawls. Reports compiled by LDWF documented 177 Gulf sturgeon incidentally captured by commercial fishers in southeastern Louisiana during 1992, of which 76 were captured in trawls, 10 in wing nets, and 91 in gillnets; LDWF noted an overall mortality rate of less than 1% (USFWS and GSMFC 1995). We also know Gulf sturgeon are occasionally captured in relocation trawls associated with dredging projects. For instance, 32 Gulf sturgeon captures were reported during relocation trawling off Alabama in 2012-2013 and 2 Gulf sturgeon captures were reported off Mississippi in 2018. Additional information regarding Gulf sturgeon bycatch is reported in Sulak et al. (2016), but quantitative estimates are still lacking due to poor observer coverage and likely low levels of self-reporting.

With only 2 observed Gulf sturgeon captures documented by our observers during 2007-2020, extrapolating out to the entire Gulf of Mexico and estimating the number of Gulf sturgeon captured specifically by the federal fishery, as was done for Atlantic sturgeon, would result in variances around the estimates and confidence intervals so large that it would render the point estimate meaningless. Application of the methods used for Atlantic sturgeon is particularly inappropriate because of differences in the species' distribution in the action area. Unlike data for Atlantic sturgeon, Gulf sturgeon data indicate that Gulf sturgeon stay primarily within nearshore waters, and records documenting their presence in federal waters are extremely limited. In order to extrapolate a reasonable CPUE, the species would have to have a more uniform distribution.

Given the low level of observer coverage (~2% observer coverage since becoming mandatory), assuming the only captures in Gulf of Mexico shrimp fisheries during the 2007-2020 period were observed is unreasonable, even though such captures are likely very rare. Based on the results of the flexible flatbar flynet TED testing, which documented 87% of Atlantic sturgeon escaped

[18] We only estimate the effect of the fisheries in federal waters, where we permit vessels and authorize shrimp fishing activity.

capture through the TED, we reasonably assumes that for every Gulf sturgeon caught in a trawl, an additional 8 may escape via TEDs (1 capture/13%=7.7). With the limited available data, we conclude that observed captures will not exceed 1 per year based on the records to date, and that an additional 8 Gulf sturgeon may interact with shrimp trawls in federal waters, but escape through a TED and be undetected.

We expect mortality of captured Gulf sturgeon to be low. We have no available information on PIM for Gulf sturgeon, but PIM risks and consequences for fish species are not the same as they are for sea turtles. While delayed mortality of Gulf sturgeon is conceivable from injuries sustained by interactions with shrimp trawls, the most significant effect pathway resulting in PIM for sea turtles is forced submergence. Because Gulf sturgeon do not breathe air, forced submergence in shrimp trawls does not present the same concern. Studies in a variety of trawl fisheries have shown that mortality of the conspecific Atlantic sturgeon incidentally caught in trawl gear is very low, with most surveys showing 0% mortality (e.g., Stein et al. 2004). Although we estimated a conservative 27% mortality rate for captured Atlantic sturgeon (see Section 5.2), this was based on 3 dead releases out of 11 captures, while both observed Gulf sturgeon captures were released alive. To be consistent, we will also utilize this higher conservative mortality rate, rounded down, and assume that 1 out of every 4 interactions will result in mortality, equating to 1 Gulf sturgeon mortality in federal waters every 4 years.

5.4 Effects to Giant Manta Ray

Research on physiological stress and post-capture mortality of giant manta ray in the southeast U.S. shrimp trawl fisheries is currently lacking, though we assume the general effects of capture (e.g., changes in blood chemistry, injury from crowding/impacts in the trawl net, air exposure following capture, etc.) are similar to those documented for other elasmobranch species (Heard et al. 2014). The impact of a capture event on an individual animal is influenced by a range of biotic and abiotic variables that can be specific to the individual (e.g., size, age, maturity and degree of physical damage) or to the type of capture event (e.g., gear type, capture duration, rapid changes in temperature and pressure and handling procedures) (Davis 2002; Skomal 2007; Frick et al. 2010a, Frick et al. 2010b; Braccini et al. 2012; Skomal and Mandelman 2012; Wilson et al. 2014). Acute stress in elasmobranchs, such as that due to fisheries capture, often results in changes in blood chemistry as energy stores (e.g., glucose) are mobilized, ion balances are disrupted and metabolites (e.g., lactate and urea) move from the muscle cells into the bloodstream (Wendelaar Bonga 1997; Skomal and Mandelman 2012). In elasmobranch species, physiological indicators of stress may not peak until hours after a stressful event, making elasmobranchs more likely to succumb to PIM caused by the accumulation of harmful metabolic byproducts at a later stage than teleost species (Frick et al. 2009). Handling and removal from the trawl net likely adds a considerable amount of additional stress, particularly for large elasmobranch species such as giant manta ray.

Estimating the Extent of Effects[19]

Carlson (2020) estimated take of giant manta ray in the shrimp trawl fisheries based on observer data. Incidental take by the fishery was estimated by the multiplication of CPUE from the observer database times the total number of trawl hours for the Gulf of Mexico by statistical grid provided by James Primrose (SEFSC, Galveston Laboratory) using methods described in Nance et al. (2008). For the South Atlantic, total shrimp trawl effort was provided by David Gloeckner (SEFSC, Miami Laboratory) using methods described in Epperly et al. (2002). In the South Atlantic, shrimp effort is reported by county and an alternate statistical grid system. Thus, the statistical grids from the shrimp fishery were overlaid with that from the South Atlantic shrimp effort data to apply the appropriate level of effort. For giant manta, total take for the Gulf of Mexico and South Atlantic was obtained by summing the take by statistical grid adjusted by the weighted CPUE. Giant manta ray were recorded captured in 2019 (n=8; Figure 9), which is the only year that recorded capture data for giant manta ray. Unfortunately, giant manta ray were not recorded to species level until 2019 (i.e., prior to 2019, all rays, including giant manta ray, were recorded together in one entry) so a comparison of long-term bycatch trends is not currently possible.

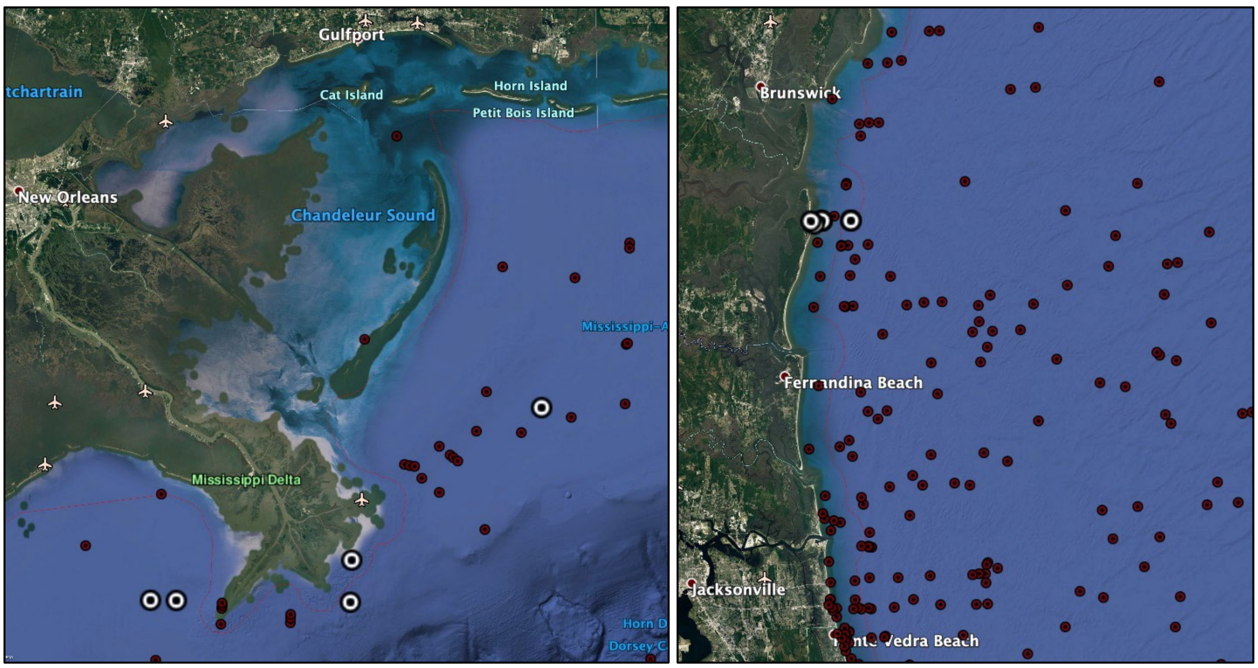

Figure 9. Locations of giant manta ray captures in the South Atlantic and Gulf of Mexico shrimp trawl fisheries based on starting tow coordinates (white circles), overlaid on locations of giant manta ray visual sighting locations (red circles).

[19] We only estimate the effect of the fisheries in federal waters, where we permit vessels and authorize shrimp fishing activity.

In 2019, the extrapolated take for giant manta ray was 1,538 and 140 animals for the South Atlantic and Gulf of Mexico, respectively (Carlson 2020). Confidence limits were also very high (i.e., 0-6,928.7) for the South Atlantic. Despite attempting to correct for giant manta ray spatial distribution and their degree of philopatry by weighing estimates of CPUE, total take estimates were extremely high, especially in the South Atlantic. In addition, the capture of several mantas in 2 tows off Georgia, combined with the low overall observer coverage, resulted in a much higher CPUE then what normally might be expected. This rate applied to the high effort in the fishery, especially off north Florida, may partially explain the high total take estimates. In the absence of higher levels of observer coverage, Carlson (2020) noted these estimates should be considered highly uncertain and most likely represent overestimates of the total bycatch. This is also true given the assumptions associated with determining total shrimp effort. Furthermore, it is very possible that some of the 8 reported captures may have been recaptures. Six of the captures occurred over 2 days on the same observed trip. The size of 4 of the 6 captured giant manta ray were estimated as 15 ft, 1 was estimated at 16 ft, and the last specimen was estimated to be 8 ft. Therefore, we expect that the species' tendency to stay in or habitually return to a particular area (i.e., philopatry) may result in higher incidence of recaptures (as compared to other organisms like sea turtles). The effect of these recaptures would exaggerate our bycatch rates on a population perspective, as well as the subsequent associated capture effects (e.g., PIM). Conversely, if recaptures of giant manta ray were occurring, it may indicate PIM-associated effects are not a significant issue for this species. We have no available information on PIM for giant manta ray, but PIM risks and consequences for fish species are not the same as they are for sea turtles. While delayed mortality of giant manta ray is conceivable from injuries sustained by interactions with shrimp trawls, the most significant effect pathway resulting in PIM for sea turtles is forced submergence. Because giant manta ray do not breathe air, forced submergence in shrimp trawls does not present the same concern. More discussion on this issue is presented below.

Methods are currently underway to improve the data and analysis related to estimating total shrimp effort and significant changes in these numbers would likely affect the incidental take estimates provided in Carlson (2020). We have no observed giant manta ray captures documented in try nets. This lack of captures may be in part due to the large size of encountered rays inhibiting their entrainment in the smaller opening (i.e., relative to standard otter trawl nets) of try nets.

Available observer data for giant manta ray captures in shrimp trawls indicate the majority of animals were released alive, though release status for 4 out of 10 animals (40%) was recorded as unknown. When an animal's condition is recorded as "unknown," it typically indicates the observer was unable to discern the animal's condition at time of release. This is not surprising given the large size of captured giant manta rays and the inability to handle or sample these large animals on the deck of a shrimp trawler. Available information indicates captured manta rays are typically released during trawl net recovery, with the animal worked out the mouth of the net at or near the surface of the water. While PIM is a potential concern for all bycatch species, there currently is no information available about PIM for giant manta ray in trawl fisheries.

Heard et al. (2014) documented overall bycatch mortality rates of 15% for smaller, benthic ray species (*Urolophus paucimaculatus*) over 3-hour trawl times. But given the larger average size (i.e., greater mass and protection of organs making it less vulnerable to injury from catch) of giant manta ray compared to the species in the aforementioned study and the fact giant manta ray respire by obligate ram ventilation versus buccal pumping of *Urolophus paucimaculatus* (i.e., buccal pumping requires a fish to continually pump water over its gills versus a ram ventilator just opening its mouth to entrain water flow, and trawling provides this water flow), we believe giant manta ray may fare relatively well following a capture event. Therefore, for the purposes of this Opinion, we believe PIM is not a significant factor for the annual estimated bycatch of giant manta ray (n=1,678 total giant manta ray) based on available information and informed judgment.

5.5 Effects to Smalltooth Sawfish

Effects of shrimp trawling in federal waters on the smalltooth sawfish are expected to result from physical interactions with fishing gear. The long toothed rostrum of the smalltooth sawfish easily entangles in the webbing of various nets, including otter trawls. Any struggle by a sawfish to escape generally results in even further entanglement as more teeth engage the webbing. Smalltooth sawfish were historically caught as bycatch in otter trawls (NMFS 2000), and early accounts document smalltooth sawfish as being frequently caught by shrimp trawls (Bigelow and Schroeder 1953b). Entangled smalltooth sawfish frequently had to be cut free, causing extensive damage to trawl nets and presenting a substantial hazard if brought on board. As a result, smalltooth sawfish caught by fishers were either killed outright or released only after removal of their saw.

Different life stages of smalltooth sawfish are associated with different habitat types and water depths. Very small and small juvenile smalltooth sawfish are most commonly associated with shallow water areas of Florida, close to shore and often associated with mangroves (Simpfendorfer and Wiley 2004). As such, these small specimens are unlikely to encounter trawls operating in federal waters of the EEZ. Large juveniles and adult smalltooth sawfish, however, are known to roam not only in shallow, inshore waters, but also occur in deeper coastal waters to depths of 200-400 ft (Poulakis and Seitz 2004) where they may be subjected to capture in shrimp trawls. While we expect most smalltooth sawfish captures to occur off Florida where the population is centered (particularly southwest Florida), there are recent anecdotal reports of smalltooth sawfish captured in trawls north of Florida.

Seitz and Poulakis (2006) list chafing and irritation of the skin, as well as the loss of rostral teeth, as consequences of entanglement in marine debris; such conditions would likely also result from trawl entanglement. Seitz and Poulakis (2006) also reported damage from incidental capture in other types of fishing gear ranging from broken rostral teeth to broken rostra. The loss of the entire rostrum or a large percentage of a rostrum would be most detrimental to an individual, as the rostrum is the primary tool in food acquisition (Wueringer et al. 2012). Depending on the extent of rostral damage, such an injury is likely to hinder an animal's ability to feed and may have long-term impacts, including mortality. Loss of rostral teeth could also cause reduced feeding efficiency as, unlike other elasmobranchs, smalltooth sawfish do not replace lost teeth (Slaughter and Springer 1968).

Estimating the Extent of Effects[20]

Carlson (2020) estimated take of smalltooth sawfish in the shrimp trawl fisheries based on observer data. Data was stratified by year, area (Gulf of Mexico or South Atlantic), and statistical grid. An estimate of uncertainty in these estimates was derived from bootstrap resampling of the calculated stratified CPUE data set. A sample was drawn from the data (with replacement) and the procedure was repeated 10,000 times to generate a mean distribution for the estimate, standard deviation, and the sample values 2.5% and 97.5% of the bootstrap distribution were used as the lower and upper bounds of the 95% confidence interval for the parameter estimate.

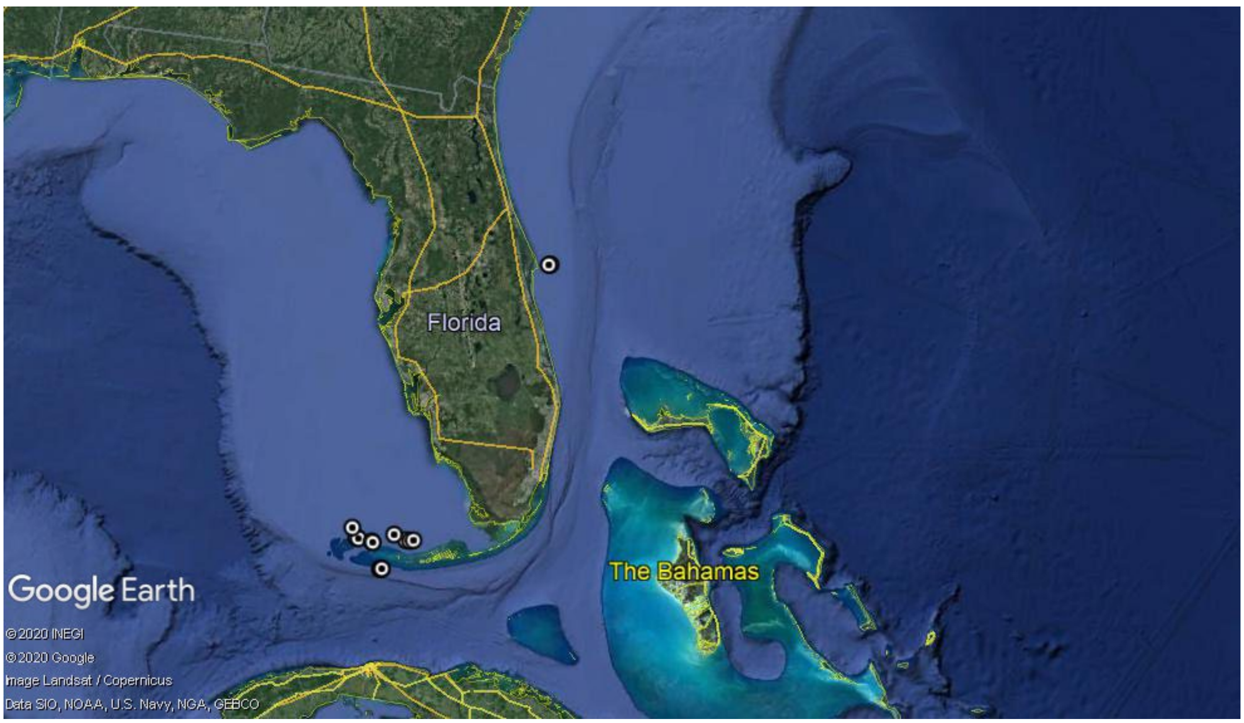

Figure 10. Locations of smalltooth sawfish captures in the South Atlantic and Gulf of Mexico shrimp trawl fisheries based on starting tow coordinates (Carlson 2020).

From January 2007 to December 2019, 17 smalltooth sawfish were recorded by observers as captured in federal waters of the Gulf of Mexico and South Atlantic by shrimp trawlers (Figure 10). Two of these sawfish, however, were captured in non-sampled tows and were excluded from further analysis. All sawfish captures occurred in statistical grids 1, 2 or 28. The

[20] We only estimate the effect of the fisheries in federal waters, where we permit vessels and authorize shrimp fishing activity.

extrapolated take (i.e., to account for low observer coverage) for smalltooth sawfish varied by year and in the Gulf of Mexico ranged from 21-331 animals and in the South Atlantic ranged from 129-207 animals; confidence limits were very high in some years. Given the inability to calculate confidence intervals in some years and statistical areas, and high confidence intervals in others (e.g., 0-4,640.8), for purposes of this Opinion we will use the mid-point (i.e., 176 and 168 for the Gulf of Mexico and South Atlantic, respectively) and as an average annual capture estimate of 344 smalltooth sawfish for the entire southeast U.S. shrimp fisheries. We have no observed smalltooth sawfish captures documented in try nets. The lack of captures may be in part due to the large size of encountered sawfish avoiding becoming entrained in the smaller opening (i.e., relative to standard otter trawl nets) of try nets.

In addition to the observer data used as the primary source of information in this effects analysis, we are aware of several additional anecdotal reports of sawfish interactions with otter trawls. Since 2015, we have documented 13 sawfish captures that were not reported by observers: 4 were captured by research trawls off Florida and Georgia; 4 captured by relocation trawls associated with dredging activities in Tampa Bay, Florida; and 5 captures by shrimp trawlers during unobserved fishing activities. Three of the research trawl captured sawfish were released in good condition, while one was released in poor condition. All of the relocation trawl captured fish were released alive, though one had a broken rostrum. Information on the 5 captures by shrimp trawlers indicate 2 were caught off South Carolina and voluntarily reported to researchers (release condition unknown), 2 were killed and were subsequently the subject of a law enforcement action, and 1 capture released alive off the coast of Florida was reported to researchers by a deckhand. While these data are not thorough enough to generate take estimates, they can provide additional insight into gear interactions and the condition or fate of sawfish upon release.

The smalltooth sawfish recovery plan (NMFS 2009b) states that available data on interactions between trawl fisheries and the U.S. DPS of smalltooth sawfish are very limited, but that shrimp trawl fisheries are associated with high sawfish mortality per interaction. As previously noted, captured sawfish may become entangled in trawl netting due to their rostrum, and their release from the net may be complicated and protracted by their large size and associated handling issues. The 2014 Opinion estimated a 36.4% bycatch mortality rate for smalltooth sawfish captured by shrimp trawls. Given the lack of information on PIM for this species, but observing our concerns associated with captured/entangled smalltooth sawfish and their subsequent handling and release during regular trawling activities (i.e., unobserved fishing), we will employee a more conservative 50% mortality estimate for captured smalltooth sawfish. This results in an annual mortality estimate of 172 smalltooth sawfish as a result of fisheries bycatch in the southeast U.S. shrimp trawl fisheries.

5.6 Summary

As discussed in Section 5.1-5.5, we anticipate the southeast U.S. shrimp fisheries to interact, capture, and potentially result in mortalities of sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish. The following tables summarize the estimated recent annual bycatch for each listed species.

Table 30. Estimates of recent annual otter trawl bycatch and mortalities in the southeast U.S. shrimp fisheries.

Species	Try Nets		Standard Nets		Total Mortalities
	Bycatch	Mortalities	Bycatch	Mortalities	
Kemp's Ridley Sea Turtle	4,212	77	5,801	440	517
Loggerhead Sea Turtle	10,603	202	3,305	120	322
Green Sea Turtle	1,095	36	2,433	208	244
Leatherback Sea Turtle	5	0	21	1	1
Hawksbill Sea Turtle	6	0	28	1	1
Atlantic Sturgeon	6	0	33	9	9
Gulf Sturgeon	-	-	1	0	0
Smalltooth Sawfish	-	-	344	172	172
Giant Manta Ray	-	-	1,678	0	0

Table 31. Estimates of recent annual otter trawl bycatch and mortalities for each Atlantic sturgeon DPS in the southeast U.S. shrimp fisheries.

DPS (MSA %)	Try Net Bycatch/Mortalities	Standard Net Bycatch/Mortalities	Total Bycatch/Mortalities
Gulf of Maine DPS (1.0%)	0/0	0/0	0/0
New York Bight DPS (3.6%)	0/0	1/0	1/0
Chesapeake Bay DPS (9.6%)	1/0	3/1	4/1
Carolina DPS (33.8%)	2/0	11/3	13/3
SA DPS (52.9%)	3/0	17/5	20/5
Atlantic Sturgeon Total[1]	6/0	33/9	39/9

[1] Note that the total bycatch and mortality of each category by DPS may be different than bycatch and mortality of Atlantic sturgeon as a whole due to rounding issues.

Table 32. Estimates of recent annual skimmer trawl bycatch and mortalities of affected sea turtles species in the southeast U.S. shrimp fisheries.

Species	Skimmer Trawl Bycatch	Skimmer Trawl Mortalities
Kemp's Ridley Sea Turtle	6,886	1,184
Loggerhead Sea Turtle	626	108
Green Sea Turtle	415	72
TOTAL	**7,928**	**1,364**

6 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area. 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 and 50 CFR 402.14.

Cumulative effects from unrelated, non-federal actions occurring in the action area may affect sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish, and their habitats. Stranding data indicate sea turtles in the action area die of various natural causes, including cold stunning and hurricanes, as well as human activities, such as incidental capture in state fisheries, ingestion of and/or entanglement in debris, ship strikes, and degradation of nesting habitat. The cause of death of most sea turtles recovered by the stranding network is unknown.

The fisheries described as occurring within the action area are expected to continue as described into the foreseeable future, concurrent with the proposed action. Numerous fisheries in state waters of the South Atlantic and Gulf of Mexico regions are known to adversely affect sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish. The past and present impacts of these activities have been discussed in Section 4 (Environmental Baseline) of this Opinion. We are not aware of any proposed or anticipated changes in these fisheries that would substantially change the impacts each fishery has on sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish covered by this Opinion.

As discussed in Section 3 and, more specifically, Section 4.4, we generally expect climate change may affect sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish, and their habitats, in a variety of ways. These changes, however, are difficult to precisely predict and slowly develop over a long period (i.e., multiple decades or longer). We have opted to use a 10-year time frame in this Opinion as a result of this uncertainty, as well as the slow transition and very gradual alteration of the environment (and subsequently on the species themselves) stemming from climate change. Yet, we do not expect to observe any climate change effects during this foreseeable time frame (i.e., 10 years) that would manifest in such a way to create a measureable risk for any species considered in this Opinion.

We did not find any information about non-federal actions other than what has already been described in Section 4 of this Opinion, most of which we expect will continue in the future. An increase in these activities could similarly increase their effect on ESA-listed species and, for some, increases in the future are considered reasonably certain to occur. Given current trends in global population growth, threats associated with climate change, pollution, fisheries bycatch, aquaculture, vessel strikes and approaches, and sound are likely to continue to increase in the future, although any increase in effect may be somewhat countered by an increase in conservation and management activities. We will continue to work with states to develop ESA Section 6 agreements and with researchers on Section 10 permits to enhance programs to quantify and mitigate these effects. Therefore, we expect that the levels of take of sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish described for each of the fisheries and non-fisheries will continue at similar levels into the foreseeable future. For the remaining activities and associated threats identified in Section 4 (including the additional discussion on climate change in Section 4.4), and other unforeseen threats, the magnitude of increase and the significance of any anticipated effects remain unknown. The best scientific and commercial data available provide little specific information on any long-term effects of these potential sources of disturbance on ESA-listed species populations. Thus, this Opinion assumes effects in the future (i.e., over the 10-year time frame) would be similar to those in the past and, therefore, are reflected in the anticipated trends described in Sections 3 and 4.

7 INTEGRATION AND SYNTHESIS OF EFFECTS

The analyses conducted in the previous sections of this Opinion provide the basis on which we determine whether the proposed action would be likely to jeopardize the continued existence of Kemp's ridley, green (NA and SA DPSs), loggerhead (NWA DPS), leatherback, and hawksbill sea turtles, as well as Atlantic sturgeon (all 5 DPSs), Gulf sturgeon, giant manta ray, and smalltooth sawfish (U.S. DPS). 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 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 actions directly or indirectly reduce 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.

Our 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. Therefore, we also evaluate in the context of the recovery plans for each species.

Recovery plans include criteria, which, when met, would result in downlisting (i.e., changing the listing from endangered to threatened) or in a determination that the species be removed from the List of Endangered and Threatened Wildlife. Recovery criteria can be viewed as targets, or values, by which progress toward achievement of recovery objectives can be measured. Recovery criteria may include such things as population numbers and sizes, management or elimination of threats by specific mechanisms, and specific habitat conditions. In newer recovery plans, recovery criteria are often framed in terms of population parameters (Demographic Recovery Criteria) and the 5 listing factors (Listing Factor Recovery Criteria). For some species, the plans have not been recently updated and do not include specific Demographic and Listing Factor Recovery Criteria. Regardless of whether these are included, we evaluate each species in the context of the criteria and objectives in its recovery plan.

The status of each listed species or DPS likely to be adversely affected by the proposed action is reviewed in Section 3. For any species listed globally, our jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery at the global species range. For any species listed as DPSs, a jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery of that DPS. Below, we re-evaluate the responses of Kemp's ridley, green (NA and SA DPSs), loggerhead (NWA DPS), leatherback, and hawksbill sea turtles, as well as Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish (U.S. DPS) to the effects of the action.

As reference, we quantified the estimated effects of the action in the recent near term on an annual basis to standardize the various available data sets amongst the ESA-listed species under consideration in this Opinion. We also noted the time frame for this Opinion is 10 years. Therefore, we will summarize the anticipated effects (i.e., captures and mortalities) to Kemp's ridley, green (NA and SA DPSs), loggerhead (NWA DPS), leatherback, and hawksbill sea turtles, as well as Atlantic sturgeon (all 5 DPSs), Gulf sturgeon, giant manta ray, and smalltooth sawfish (U.S. DPS) that we expect will occur over the next 10 years. The annual effects of the action for each ESA-listed species and species' DPS discussed in Section 5 have been extrapolated out for this 10-year time period, and are presented in Tables 33-36 below.

To project the effects of the proposed action into the near future and over the 10-year time frame of this Opinion, we also need to consider potential changes in both the fisheries (e.g., effort) and the affected species (i.e., population changes) to what was estimated in Section 5. In regards to the fisheries, we do not expect any substantial increases in participation and effort in the foreseeable future, however, we do anticipate potential near-term decreases in effort as a result of COVID-19 that may manifest in 2020 and through early 2021. When looking at recent population trends of affected species, we do note species that may be experiencing population growth to a point that future interactions may diverge from our recent "static" estimates of bycatch and bycatch mortality. That is, there is a possibility that increased population growth could lead to greater geographical distribution and/or densities in areas where the fisheries are prosecuted, resulting in increased bycatch, even with constant fisheries effort. For sea turtles, we use recent (i.e., 10-year) nesting trends as a recognized population proxy to determine if population growth may impact our estimates into the near future. Nesting for some species, such as loggerhead sea turtles, demonstrate positive changes in numbers, however, the recent trend is not statistically significant. Kemp's ridley sea turtles demonstrate significant increases in nesting numbers (and assumed overall population increases) over a longer time frame (e.g., 25 years), but trends are more unstable and less clear in the more recent 10-year time frame. Thus, we make no population growth adjustments for those species.

Green sea turtles, though, do demonstrate a general increase in nesting numbers over the past 10 years. Based on the nesting data presented in Section 3, we estimate the 10-year annualized green sea turtle population growth rate to be 15.8% from 2010-2019, though some perspective and caution is needed with this estimate. First, green sea turtles exhibit a bi-annual nesting trend and the end of our utilized time frame (i.e., 2019) is a high peak in that trend. In 2010, however, the beginning of our time frame did not coincide with the usual high-low pattern, as it was higher than 2009 but lower than 2011. Therefore, we believe using the more recent 5-year annualized nesting increase estimate of 7.6% is more appropriate. Caution is still needed when using this estimate, however, as the recent increases in nesting may slow as carrying capacity or other population bottlenecks are reached. Available information indicates that fisheries bycatch may have a disproportionate impact on smaller juvenile and sub-adult green sea turtles in the shrimp fisheries. Therefore, we believe utilizing nesting information as insight into potential population increases is a valid metric when estimating the potential effects of the fisheries into the future. As such, we will apply a 7.6% increase into our 10-year estimates of bycatch for green sea turtles in Tables 33 and 35-36 below. We then use the bycatch mortality relationship gleaned from existing mortalities/bycatch in Tables 30 and 32 in Section 5 to modify the mortality estimates for green sea turtles. For example, we extrapolate out the annual 2,433 green sea turtle bycatch estimate for standard nets in Table 30 for our 10-year time frame to obtain 24,330 green sea turtles over the next 10 years. We then modify by 7.6% to reflect expected population growth, which results in an adjusted 26,179 green sea turtles (2,433 x 10 = 24,330 x 0.076 = 1,849.08 +

24,330 = 26,179.08). The final step is to adjust for bycatch mortality, which is 8.5491% based on a direct relationship in Table 30 (208 / 2,433 = 0.085491). Using this rate, we calculate a new adjusted mortality of 2,238 green sea turtles over the following 10-year time period, which we use for the following jeopardy analysis in Section 7 (Integration and Synthesis of Effects) and when authorizing take in Section 8 of this Opinion.

Likewise, available information indicates smalltooth sawfish is undergoing recovery and their population may be increasing in number and distribution. Carlson et al. (2007) noted that yearly estimates of abundance indicate that smalltooth sawfish may be increasing within Everglades National Park at a rate of about 5% per year, though confidence limits were large (i.e., 0-10% per year). Yet, this rate is similar to intrinsic rates of increase for smalltooth sawfish of 8-13% per year calculated using a demographic approach (Simpfendorfer 2000). We believe using a 5% population increase rate is appropriate to properly evaluate the effects of the proposed action on the species and is, in fact, a conservative approach. That is, if one were to under-estimate incidental bycatch on the species, the corresponding jeopardy analysis would be skewed to under-represent effects on the species. Therefore, this 5% increase has been incorporated into our 10-year estimates of bycatch of smalltooth sawfish presented in Table 33. To be specific, the unadjusted 10-year estimate of 3,440 captures of smalltooth sawfish was increased to 3,612 (3,440 x 0.05 = 172 + 3,440 = 3,612). We then use the 50% mortality estimate discussed in Section 5.5 to calculate adjusted mortalities as 1,806 individuals.

Table 33. Estimates of southeast U.S. shrimp fisheries otter trawl bycatch and mortality over the next 10 years. Rows highlighted in red indicate species (i.e., green sea turtle and smalltooth sawfish) with adjusted capture and mortality estimates that take into consideration anticipated population growth.

Species	Try Nets		Standard Nets		Total Mortalities
	Captures	Mortalities	Captures	Mortalities	
Kemp's Ridley Sea Turtle	42,120	770	58,010	4,400	5,170
Loggerhead Sea Turtle	106,030	2,020	33,050	1,200	3,220
Green Sea Turtle	11,782	388	26,179	2,238	2,626
NA DPS (96%)	11,310	372	25,132	2,148	2,520
SA DPS (4%)	472	16	1,048	90	106
Leatherback Sea Turtle	50	0	210	10	10
Hawksbill Sea Turtle	60	0	280	10	10
Atlantic Sturgeon	60	0	330	90	90
Gulf Sturgeon	-	-	10	2	2
Smalltooth Sawfish	-	-	3,612	1,806	1,806
Giant Manta Ray	-	-	16,780	0	0

Table 34. Estimates of southeast U.S. shrimp fisheries otter trawl bycatch and mortality on each Atlantic sturgeon DPS over the next 10 years.

DPS (MSA %)	Try Net Bycatch/Mortalities	Standard Net Bycatch/Mortalities	Total Bycatch/Mortalities
Gulf of Maine DPS (1.0%)	0/0	4/0	4/0
New York Bight DPS (3.6%)	2/0	12/4	14/4
Chesapeake Bay DPS (9.6%)	6/0	32/8	38/8
Carolina DPS (33.8%)	20/0	112/30	132/30
SA DPS (52.9%)	32/0	174/48	206/48
Atlantic Sturgeon Total[1]	60/0	330/90	390/90

[1] Note that the total bycatch and mortality of each category by DPS may be different than bycatch and mortality of Atlantic sturgeon as a whole due to rounding issues.

Table 35. Estimates of southeast U.S. shrimp fisheries skimmer trawl bycatch and bycatch mortality over the next 10 years. Rows highlighted in red indicate species (i.e., green sea turtle) with adjusted capture and mortality estimates that take into consideration anticipated population growth.

Species	Skimmer Trawl Captures	Skimmer Trawl Mortalities
Kemp's Ridley Sea Turtle	68,860	11,840
Loggerhead Sea Turtle	6,260	1,080
Green Sea Turtle	4,466	774
NA DPS (96%)	4,288	744
SA DPS (4%)	178	30

Table 36. Estimates of total (otter and skimmer trawl gear, all nets combined) sea turtle bycatch and bycatch mortality in the southeast U.S. shrimp fisheries over the next 10 years. Rows highlighted in red indicate species (i.e., green sea turtle) with adjusted capture and mortality estimates that take into consideration anticipated population growth.

Species	Captures	Mortalities
Kemp's Ridley Sea Turtle	168,990	17,010
Loggerhead Sea Turtle	145,340	4,300
Green Sea Turtle	42,428	3,400
NA DPS (96%)	40,730	3,264
SA DPS (4%)	1,698	136
Leatherback Sea Turtle	260	10
Hawksbill Sea Turtle	340	10

7.1 Kemp's Ridley Sea Turtle

Concentrated in the shallow waters of the Gulf of Mexico and Atlantic coast where shrimp fisheries pressure is also concentrated, we expect Kemp's ridley sea turtles to be the species most affected by the proposed action. We estimate that the proposed action will result in a total of 168,990 captures and 17,010 mortalities of Kemp's ridley sea turtles over the next 10 years. The nonlethal capture of 151,980 Kemp's ridley sea turtles (168,990 captures - 17,010 mortalities from direct effects and PIM = 151,980 nonlethal captures) over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Either male or female Kemp's ridleys may be captured in the shrimp fisheries since available information suggests that both sexes occur in the action area. We anticipate, however, that a significant portion of the Kemp's ridleys interacting with the shrimp fisheries in the action area are expected to be sexually immature juvenile specimens based on available observer data. Furthermore, we are requiring 3-in bar spacing for skimmer trawl vessels 40 ft and greater in length specifically because of the small size of encountered Kemp's ridley sea turtles.

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 DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this DPS. We also included an extensive section on Climate Change in Section 4.4. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. These effects are in addition to the other ongoing effects to the species, such as bycatch in other fisheries, effects from other federal actions, and the potential effects of climate change, all of which were already discussed in detail in the preceding sections of this biological opinion. It is important to note that virtually all of the effects discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with other sea turtles species, nest count data must be interpreted with caution given that these estimates provide a minimum count of the number of nesting Kemp's ridley sea turtles. In addition, the estimates do not account for adult males or juveniles of either sex. Without information on the proportion of adult males to females and the age structure of the population, nest counts cannot be used to estimate the total population size (Meylan 1982; Ross 1996). Nevertheless, the nesting data does provide valuable information on the extent of Kemp's ridley nesting and the trend in the number of nests laid, and represents 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, unpublished data). In 2013 and 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 nests, and in 2016 overall numbers increased to 18,354 recorded nests. 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 as cited in NMFS 2020b) and decreases observed in 2018 and again in 2019. In 2019, there were 11,140 documented nests in Mexico. It is unknown whether this decline is related to resource fluctuation, natural population variability, effects of catastrophic events like the DWH oil spill event affecting the nesting cohort, or some other factor(s). A small nesting population is also emerging in the United States, primarily in Texas. From 1980-1989, there were an average of 0.2 nests/year at PAIS, rising to 3.4 nests/year from 1990-1999, 44 nests/year from 2000-2009, and 110 nests per year from 2010-2019. There was a record high of 353 nests in 2017 (NPS 2020). 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 (NMFS 2020b) and decreases in nesting in 2018 and 2019 (NPS 2020).

Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (NMFS and USFWS 2015; TEWG 2000). Galloway et al. (2016) developed a stock assessment model for Kemp's ridley to evaluate the relative contributions of conservation efforts and other factors toward this species' recovery. Terminal population estimates for 2012 summed over ages 2 to 4, ages 2+, ages 5+, and ages 9+ suggest that the respective female population sizes were 78,043 (SD = 14,683), 152,357 (SD = 25,015), 74,314 (SD =10,460), and 28,113 (SD = 2,987) (Gallaway et al. 2016). Using the standard IUCN protocol for sea turtle assessments, the number of mature individuals was recently estimated at 22,341 (Wibbels and Bevan 2019). The calculation took into account the average annual nests from 2016-2018 (21,156), a clutch frequency of 2.5 per year, a remigration interval of 2 years, and a sex ratio of 3.17 females:1 male. Based on the data in their analysis, the assessment concluded the current population trend is unknown (Wibbels and Bevan 2019). However, some positive outlooks for the species include recent conservation actions, including the expanded TED requirements in the skimmer trawl sector of the shrimp fisheries (84 FR 70048, December 20, 2019; 86 FR 16676, March 31, 2021) and a decrease in the amount of overall shrimping off the coast of Tamaulipas and in the Gulf of Mexico (NMFS and USFWS 2015).

Genetic variability in Kemp's ridley turtles is considered to be high, as measured by nuclear DNA analyses (i.e., microsatellites) (NMFS et al. 2011). If this holds true, then rapid increases in population over one or two generations would likely prevent any negative consequences in the genetic variability of the species (NMFS et al. 2011). Additional analysis of the mtDNA taken from samples of Kemp's ridley turtles at Padre Island, Texas, showed six distinct haplotypes, with one found at both Padre Island and Rancho Nuevo (Dutton et al. 2006).

Kemp's ridleys mature and nest at an age of 7-15 years, which is earlier than other sea turtles. A younger age at maturity may be a factor in the response of this species to recovery actions. The required use of TEDs in shrimp trawls in the United States under the sea turtle conservation regulations and in Mexican waters as required by their federal regulations has had dramatic effects on the recovery of Kemp's ridley sea turtles. Kemp's ridley sea turtles total mortality (all sources) declined by about one-third with the early implementation of TEDs, and it has been estimated that after 1996 mortality declined by almost 60% compared to pre-TED levels.

The proposed action 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. Using the estimate of mature animals in Wibbels and Bevan (2019), the loss of 17,010 animals over a 10-year period represents would represent an approximate 7.6% reduction of the overall sexually-mature population. Based on average size of captured Kemp's ridleys documented by fishery observers in the shrimp fisheries, however, we know a significant portion of these turtles are small, sexually-immature juvenile sea turtles, many of which would not survive to reach maturity and reproduce. As a result, we believe the reduction in the overall Kemp's ridley sea turtle population (immature and mature) is much less significant considering sea turtle species are expected to have increasing numbers of specimens when looking at overall population by descending age (i.e., older to younger). The proposed action could also result in a potential reduction in future reproduction, assuming at least some of these individuals would be female and would have survived to reproduce in the future. The annual loss of adult females could preclude the production of thousands of eggs and hatchlings, of which a small percentage is expected to survive to sexual maturity. Thus, the death of any females that would otherwise have survived to sexual maturity would eliminate their contribution to future generations, and result in a reduction in sea turtle reproduction. The anticipated lethal interactions are expected to occur anywhere in the action area, and sea turtles generally have large ranges in which they disperse. Thus, no reduction in the distribution of Kemp's ridley sea turtles is expected from fisheries interactions. Whether the reductions in numbers and reproduction of Kemp's ridley sea turtles would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In addition, the species' limited range and low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental stochasticity, which are often difficult to predict with any certainty.

It is likely that the Kemp's ridley was the sea turtle species most affected by the DWH oil spill event on a population level. In addition, sea turtle strandings documented from 2010 to present in Alabama, Louisiana, and Mississippi primarily involved Kemp's ridley sea turtles. Necropsy results indicated that a significant proportion of turtle mortality was caused by forced submergence, which is commonly associated with fishery interactions (77 FR 27413, May 10, 2012). As described in Section 4 (Environmental Baseline), regulatory actions have been taken to reduce anthropogenic effects to Kemp's ridley sea turtles. These include measures implemented to reduce the number and severity of Kemp's ridley sea turtle interactions in the Mid-Atlantic large mesh gillnet, Mid-Atlantic summer flounder, Mid-Atlantic scallop dredge, and the Virginia pound net fisheries. In 2021, TED requirements in a portion of the skimmer trawl sector of the shrimp fisheries will become effective, further reducing impacts to sea turtles.

There are no new known sources of mortality for Kemp's ridley sea turtles other than potential impacts from the DWH oil spill event, and highly uncertain potential future impacts associated with climate change. As mentioned in previous sections, some of the likely effects commonly associated with climate change are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The potential effects, and the expected related effects to ESA-listed species (e.g., impacts to sea turtle nesting beaches and hatchling sex ratios, associated effects to prey species, etc.) stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the Kemp's ridley sea turtle population. Furthermore, the effects on Kemp's ridley sea turtles from the proposed actions are not likely to appreciably reduce overall population numbers over time due to current population size, expected recruitment, and the implementation of additional conservation requirements in the shrimp trawl fisheries, even in light of the adverse impacts expected to have occurred from the DWH oil spill event.

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 nesting data, the population trend has become less clear. Nonetheless, data from 1990 to present continue to support that Kemp's ridley sea turtles have shown a generally increasing nesting trend. Even with reported biennial fluctuations in nesting numbers from Mexican beaches, all years since 2006 have reported over 10,000 nests per year, indicating an increasing population over the previous decades. We believe this long-term trend in nesting is likely evidence of a generally increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. These nesting data are indicative of a species with a high number of sexually mature individuals. All of those positive population trends have arisen while the shrimp fisheries have been operating and adversely affecting the species along with all the other adverse effects included in the baseline. The loss of 17,010 Kemp's ridleys over a 10-year period is not expected to change the trend in nesting, the distribution of, or the reproduction of Kemp's ridley sea turtles in the next 10 years or beyond. 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 Kemp's ridley sea turtle (NMFS et al. 2011) lists the following recovery objectives for downlisting that are relevant to the fisheries assessed in this Opinion:

- Demographic: A population of at least 10,000 nesting females in a season (as measured by clutch frequency per female per season) distributed at the primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) in Mexico is attained. Methodology and capacity to implement and ensure accurate nesting female counts have been developed.
- Listing factor: TED regulations, or other equally protective measures, are maintained and enforced in U.S. and Mexican trawl fisheries (e.g., shrimp, summer flounder, whelk) that are known to have an adverse impact on Kemp's ridleys in the Gulf of Mexico and Northwest Atlantic Ocean.

With respect to the demographic recovery objective, the nesting numbers in the most recent three years indicate there were 24,570 nests in 2017, 17,945 in 2018, and 11,090 in 2019 on the main nesting beaches in Mexico. Based on 2.5 clutches/female/season, these numbers represent approximately 9,828 (2017), 7,178 (2018), and 4,436 (2019) nesting females in each season. The number of nests reported annually from 2010 to 2014 declined overall; however, they rebounded in 2015 through 2017, and declined again in 2018 and 2019. Although there has been a substantial increase in the Kemp's ridley population within the last few decades, the number of nesting females is still below the number of 10,000 nesting females per season required for downlisting (NMFS and USFWS 2015). Since we concluded that the potential loss of Kemp's ridley sea turtles is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede 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.

In regards to the listing factor recovery criterion, the recovery plan states, "the highest priority needs for Kemp's ridley recovery are to maintain and strengthen the conservation efforts that have proven successful. In the water, successful conservation efforts include maintaining the use of TEDs in fisheries currently required to use them, expanding TED-use to all trawl fisheries of concern, and reducing mortality in gillnet fisheries. Adequate enforcement in both the terrestrial and marine environment also is also noted essential to meeting recovery goals" (NMFS et al. 2011). We are currently expanding the use of TEDs in skimmer trawls, which should aid in the recovery of the species. The required use of TEDs in shrimp trawls in the United States under sea turtle conservation regulations and in Mexican waters has had dramatic effects on the recovery of Kemp's ridley sea turtles.

In summary, the loss of 17,010 Kemp's ridley sea turtles over the next 10 years and beyond[21] as a result of the fisheries considered in this Opinion—even amidst other ongoing threats to the

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[21] As initially discussed in Section 2.1, the lifespan (i.e., ESA coverage) for the proposed action covered in this Opinion is 10 years. We recognize, however, the proposed action (i.e., the southeast U.S. shrimp fisheries) will continue into the future. Moreover, the effects from the proposed action are not strictly instantaneous and occurring only within the 10-year lifespan of this Opinion; loss of nesting female sea turtles, for instance, can have population "echoes" for several generations into the future. As such, we acknowledge these effects with the addition of "and beyond" language in our jeopardy analyses.

species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or the potential effects of climate change—will not appreciably reduce the likelihood of survival and recovery for Kemp's ridley sea turtles given the long term nesting trend, the population size, and ongoing and future measures (e.g., expanded TED regulations in the shrimp trawl fishery) that reduce the number of Kemp's ridley sea turtles that are injured and die.

7.2 Green Sea Turtle

As noted in Section 3, we anticipate green sea turtles within the action area affected by the proposed action would consist of 96% from the NA DPS and 4% from the SA DPS based on the majority of fishery effort occurring in the Gulf of Mexico. We provide separate jeopardy analyses for each DPS below based on this DPS percentage split.

7.2.1 Green Sea Turtle NA DPS

We estimate that the proposed action will result in a total of 40,730 captures that will result in 3,264 mortalities of green sea turtles (NA DPS) over the next 10 years. The nonlethal capture of 37,466 green sea turtles (40,730 captures - 3,264 mortalities from direct effects and PIM = 37,466 nonlethal captures) from the NA DPS over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Nonlethal captures will not result in a reduction in numbers of the species, as we anticipate these nonlethal captures to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur anywhere within the action area and would be released within the same general area where caught, we anticipate no change in the distribution of NA DPS green sea turtles. The mortality of 3,264 green sea turtles from the NA DPS over 10 years is an obvious reduction in numbers. These moralities could also result in a potential reduction in future reproduction, assuming some individuals would be female and would have survived to reproduce in the future. For example, an adult green sea turtle can lay 3-4 clutches of eggs every 2-4 years, with approximately 110-115 eggs/nest, of which a small percentage are expected to survive to sexual maturity. These mortalities are anticipated to occur over the large area of the action area, however, green sea turtles in the NA DPS generally have large ranges; thus, no reduction in the distribution is expected from these mortalities.

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 DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this DPS. We also included an extensive section on Climate Change in Section 4.4. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. These effects are in addition to the other ongoing effects to the species, such as bycatch in other fisheries, effects from other federal actions, and the potential effects of climate change, all of which were already discussed in detail in the preceding sections of this biological opinion. It is important to note that virtually all of the effects discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

Seminoff et al. (2015) estimated that there are greater than 167,000 nesting green sea turtle females in the NA 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, U.S. (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 2010 increased, despite substantial human impacts to the population at the nesting beach and at foraging areas (Campell and Lagueux 2005; Troëng and Rankin 2005). Nesting locations in Mexico along the Yucatan Peninsula also indicate the number of nests laid each year has 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, *El Centro De Investigaciones De Quintana Roo*, unpublished data, 2013, in Seminoff et al. 2015). In Florida, most nesting occurs along the eastern central Atlantic coast, 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, FWC, pers. comm., 2013). As described in Section 3 of this Opinion, nesting has increased substantially over the last 20 years and peaked in 2017 with 53,102 nests statewide in Florida, though the number of nests dropped again in 2018 as part of the regular biennial fluctuation.

Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of green sea turtles would not be appreciably affected. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals. Since the abundance trend information for green sea turtles is clearly increasing while mortalities have been occurring, we believe the mortalities attributed to the proposed action will not have any measurable effect on that trend. In addition, 3,264 green sea turtles over 10 years represents a very small fraction (<0.2% annually) of the overall NA DPS female nesting population estimated by Seminoff et al. (2015). As described in Section 4, although the DWH oil spill event is expected to have resulted in adverse impacts to green sea turtles, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from these fisheries would result in a detectable change in the population status of green sea turtles in the Atlantic. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions could be seen as reducing the likelihood of survival and recovery of the species.

As mentioned in previous sections, some of the likely effects commonly associated with climate change are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The potential effects, and the expected related effects to ESA-listed species (e.g., impacts to sea turtle nesting beaches and hatchling sex ratios, associated effects to prey species, etc.) stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the NA DPS green sea turtle population.

In summary, green sea turtle nesting at the primary nesting beaches within the range of the NA DPS has been increasing over the past 2 decades, against the background of the past and ongoing human and natural factors (i.e., the environmental baseline) that have contributed to the current status of the species. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for NA DPS green sea turtles is increasing, we believe 37,466 nonlethal captures and the mortality of 3,264 green sea turtles over a 10-year period considered by this Opinion will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action covered under this Opinion is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle NA DPS in the wild.

As described in Section 4 (Environmental Baseline), regulatory actions have been taken to reduce anthropogenic effects to green sea turtles in the Atlantic. These include measures to reduce the number and severity of green sea turtle interactions in other fisheries, such as the Mid-Atlantic large mesh gillnet, Mid-Atlantic sea scallop dredge, summer flounder trawl, and the Virginia pound net fisheries—all of which are causes of green sea turtle mortality in the Atlantic. Since most of these regulatory measures have been in place for several years now, it is likely that current nesting trends reflect the benefit of these measures to Atlantic green sea turtles. Therefore, the current nesting trends for green sea turtles in the Atlantic are likely to continue to improve as a result of the regulatory actions taken for these and other fisheries. There are no new known sources of mortality for green sea turtles in the Atlantic other than potential impacts from the DWH oil spill event.

The recovery plan for Atlantic green sea turtles (NMFS and USFWS 1991) lists the following recovery objectives, which are relevant to the proposed action in this Opinion, and must be met over a period of 25 continuous years:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least six years;
- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

Along the Atlantic coast of eastern central Florida, 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, FWC, pers. comm., 2013, as cited in Seminoff et al. 2015). Nesting has increased substantially over the last 20 years and peaked in 2011 with 15,352 nests statewide (Chaloupka et al. 2007; B. Witherington, FWC, pers. comm., 2013 as cited in Seminoff et al. 2015). The status review estimated total nester abundance for Florida at 8,426 turtles (Seminoff et al. 2015). As described above, sea turtle nesting in Florida is increasing. For the most recent 6-year period of statewide nesting beach survey data, there were 5,895 nests in 2014, 37,341 in 2015, 5,393 in 2016, 53,102 in 2017, 4,545 in 2018, and 53,011 in 2019 (see https://myfwc.com/research/wildlife/sea-turtles/nesting/beachsurvey-totals/). Thus, this recovery criterion continues to be met.

Several actions are being taken to address the second objective; however, there are currently few studies, and no estimates, available that specifically address changes in abundance of individuals on foraging grounds. A study in the central region of the Indian River Lagoon (along the east coast of Florida) found a 661% increase in juvenile green sea turtle capture rates over a 24-year study period from 1982-2006 (Ehrhart et al. 2007). Wilcox et al. (1998) found a dramatic increase in the number of green sea turtles captured from the intake canal of the St. Lucie nuclear power plant on Hutchinson Island, Florida beginning in 1993. During a 16-year period from 1976-1993, green sea turtle captures averaged 24 per year. Green sea turtle catch rates for 1993, 1994, and 1995 were 745%, 804%, and 2,084% above the previous 16-year average annual catch rates (Wilcox et al. 1998). In a study of sea turtles incidentally caught in pound net gear fished in inshore waters of Long Island, New York, Morreale and Standora. (2005) documented the capture of more than twice as many green sea turtles in 2003 and 2004 with less pound net gear fished, compared to the number of green sea turtles captured in pound net gear in the area during the 1990s. Yet other studies have found no difference in the abundance (decreasing or increasing) of green sea turtles on foraging grounds in the Atlantic (Bjorndal et al. 2005; Epperly et al. 2007). Given the clear increases in nesting, however, it is reasonably likely that numbers on foraging grounds have increased.

We conclude the loss of 3,264 green sea turtles over the next 10 years and beyond as a result of the fisheries considered in this Opinion—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—

will not appreciably reduce the likelihood of survival for green sea turtles. This conclusion is based on the above findings where we demonstrated the number of mortalities are not expected to measurably affect the increasing nesting trend in Florida, that the population size is relatively large, and that we have implemented other conservation measures to reduce the number of NA DPS green sea turtle mortalities, which should result in increases to the numbers of NA DPS green sea turtles that would otherwise not have occurred in the absence of those regulatory measures. Given the proposed action is not expected to measurably affect nesting trends, it will also not appreciably reduce the likelihood of recovery of NA DPS green sea turtles. Therefore, we conclude the proposed action considered in this Opinion is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NA DPS of green sea turtles in the wild.

7.2.2 Green Sea Turtle SA DPS

We estimate that the proposed action will result in a total of 1,698 captures and 136 mortalities of green sea turtles (SA DPS) over the next 10 years. The nonlethal capture of 1,562 green sea turtles (1,698 captures - 136 mortalities from direct effects and PIM = 1,562 nonlethal captures) from the SA DPS over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Nonlethal captures will not result in a reduction in numbers of the species, as we anticipate these nonlethal captures to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur anywhere within the action area and would be released within the same general area where caught, we anticipate no change in the distribution of NA DPS green sea turtles. The mortality of 136 green sea turtles from the SA DPS over 10 years is an obvious reduction in numbers. As discussed in Section 7.2.1, these moralities could also result in a potential reduction in future reproduction, assuming some individuals would be female and would have survived to reproduce in the future. These mortalities are anticipated to occur over the large area of the action area, however, green sea turtles in the SA DPS generally have large ranges; thus, no reduction in the distribution is expected from these mortalities.

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 DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this DPS. We also included an extensive section on Climate Change in Section 4.4. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. These effects are in addition to the other ongoing effects to the species, such as bycatch in other fisheries, effects from other federal actions, and the potential effects of climate change, all of which were already discussed in detail in the preceding sections of this biological opinion. It is important to note that virtually all of the effects discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

The SA DPS is large, estimated at over 63,000 nesting females, but data availability is poor with 37 of the 51 identified nesting sites not having sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). 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. Some of the largest nesting beaches such as Ascension and Aves Islands in Venezuela and Galibi in Suriname appear to be increasing, with others (e.g., Trindade and Atol das Rocas, Brazil; Poilão and the rest of Guinea-Bissau) appearing to be stable. In the U.S., nesting of SA DPS green sea turtles occurs in the SA DPS on beaches of the U.S. Virgin Islands, primarily on Buck Island and Sandy Beach, St. Croix, although there are not enough data to establish a trend. We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of green sea turtles from the SA DPS in the wild. Although the potential mortality of 136 sea turtles from this DPS over a 10-year period may occur as a result of the proposed action and would result in a reduction in absolute population numbers, the population of green sea turtles in the SA DPS would not be appreciably affected. Likewise, the reduction in reproduction that could occur due to these mortalities would not appreciably affect reproduction output in the South Atlantic.

As mentioned in previous sections, some of the likely effects commonly associated with climate change are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The potential effects, and the expected related effects to ESA-listed species (e.g., impacts to sea turtle nesting beaches and hatchling sex ratios, associated effects to prey species, etc.) stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the SA DPS green sea turtle population.

As discussed for the NA DPS, the recovery plan for Atlantic green sea turtles (NMFS and USFWS 1991) lists the following recovery objectives, which are relevant to the proposed action in this Opinion, and must be met over a period of 25 continuous years:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least six years;
- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

The nesting recovery objective is specific to the NA DPS, but demonstrates the importance of increases in nesting to recovery. As previously stated, nesting at the primary SA DPS nesting beaches has been increasing over the past 3 decades. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting and in-water abundance, however, it is likely that numbers on foraging grounds have increased.

The potential for 136 green sea turtle mortalities from the SA DPS over the next 10 years and beyond will result in a reduction in numbers when they occur, but it is unlikely to have any detectable influence on the trends noted above, even when considered in context with the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. Similarly, we do not expect the nonlethal capture of 1,562 green sea turtles from the SA DPS to have any detectable influence on the recovery objectives. Therefore, we conclude the proposed action considered in this Opinion—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of green sea turtles in the wild.

7.3 Loggerhead Sea Turtle NWA DPS

We estimate that the proposed action will result in a total of 145,340 captures and 4,300 mortalities of loggerhead sea turtles over the next 10 years. The nonlethal capture of 141,040 loggerhead sea turtles (145,340 captures - 4,300 mortalities from direct effects and PIM = 141,040 nonlethal captures) from the NWA DPS over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Nonlethal captures will not result in a reduction in numbers of the species, as we anticipate these nonlethal captures to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur anywhere within the action area and would be released within the same general area where caught, we anticipate no change in the distribution of NA DPS green sea turtles. The mortality of 4,300 loggerhead sea turtles from the NWA DPS due to the proposed action occurring over a 10-year period will reduce the number of loggerhead sea turtles compared to the number that would have been present in the absence of the proposed actions (assuming all other variables remained the same). These lethal interactions would also result in a future reduction in reproduction due to lost reproductive potential, as some of these individuals would be females who would have reproduced in the future, thus eliminating each female individual's contribution to future generations. For example, an adult female loggerhead sea turtle in the NWA DPS can lay 3 or 4 clutches of eggs every 2 to 4 years, with 100 to 126 eggs per clutch (NMFS and USFWS 2008). The annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. A reduction in the distribution of loggerhead sea turtles is not expected from lethal interactions attributed to the proposed action. Because all the potential interactions are expected to occur at random throughout the action area and loggerheads generally have large ranges in which they disperse, the distribution of loggerhead sea turtles in the action area is expected to be unaffected.

Whether the reductions in loggerhead sea turtle numbers and reproduction as a result of 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. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects are to the extent that adverse effects on population dynamics are appreciable. In Section 3 (Status of the Species), we reviewed the status of the NWS DPS of loggerhead sea turtles in terms of nesting, female population trends, and several of the most recent assessments based on population modeling. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. Those actions include the Atlantic Pelagic Longline fishery and Gulf Reef Fish fishery, among many others, which are known to interact with the species. We also included an extensive section on Climate Change in Section 4.4. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends. 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).

NMFS (2009f) estimated the minimum adult female population size for the NWA DPS[22] in the 2004-2008 time frame 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 NWA DPS 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 NWA DPS was also obtained, with a likely range of approximately 30,000-300,000 individuals, up to less than

[22] We refer to the NWA DPS, even when discussing information in references published prior to the 2011 DPS listing, for consistency and ease of interpretation in this analysis.

1,000,000. NMFS (2011a) preliminarily estimated the loggerhead population in the NWA DPS 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. Our NEFSC's point estimate increased to approximately 801,000 individuals when including data on unidentified sea turtles that were likely loggerheads. NMFS (2011a) 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 fourth 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).

We have not previously conducted a population viability analysis (PVA) for the NWA DPS of loggerhead sea turtles in the southeast U.S., and opted again not to conduct one for this Opinion. While we have utilized a PVA for loggerheads in some capacity for other fisheries (e.g., the Atlantic sea scallop fishery, though that analysis did not model the viability of the entire loggerhead population), we ultimately decided not to pursue a PVA for this action as a PVA for the NWA DPS of loggerheads, or any other DPS for that matter, has not been constructed since there are no estimates of the number of mature males, immature males, and immature females in the population and the age structure of the population is unknown. The approach employed in this Opinion is consistent with past analyses conducted on this and other fisheries in the southeast U.S., and we believe its conclusions are sound and accurate.

In summary, abundance estimates accounting for only a subset of the entire loggerhead sea turtle population in the NWA DPS indicate the population is large (i.e., several hundred thousand individuals). Furthermore, overall long-term nesting trends have been level or increasing over the years.

The proposed action could remove up to 4,300 individuals over a 10-year period, or an annual average of 430 loggerhead sea turtles. These removed individuals represent approximately 0.11% annually on the low end of the NMFS (2011a) estimate of 381,941 loggerheads within the Northwest Atlantic continental shelf (as opposed to pelagic juveniles on the open ocean). As noted above, this estimate reflects a subset of the entire population for the NWA DPS of loggerhead sea turtles, and thus these individuals represent an even smaller proportion of the population removed. While the loss of 4,300 individuals over 10 years is an impact to the population, in the context of the overall population's size and current trend, we do not expect it 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 population within the NWA DPS to remain large (i.e., hundreds of thousands of individuals) and to retain the potential for recovery. We also expect the proposed action will 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. Therefore, we conclude the proposed action is not likely to appreciably reduce the likelihood of the NWA DPS of loggerhead sea turtles' survival in the wild.

As described in Section 4, we believe that the DWH oil spill event had an adverse impact on loggerhead sea turtles, and resulted in mortalities to an unquantified number of individuals, along with unknown lingering impacts resulting from nest relocations, nonlethal exposure, and foraging resource impacts. However, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from Southeast shrimp fisheries would result in a detectable change in the population status of the NWA DPS of loggerhead turtles. This is especially true given the size of the population and that, unlike Kemp's ridleys, the NWA DPS is proportionally much less intrinsically linked with the Gulf of Mexico. It is possible that the DWH oil release event reduced that survival rate of all age classes to varying degrees, and may continue to do so for some undetermined time into the future. However, there is no information at this time that it has, or should be expected to have, substantially altered the long-term survival rates in a manner that would significantly change the population dynamics compared to the conservative estimates used in this Opinion.

As mentioned in previous sections, some of the likely effects commonly associated with climate change are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The potential effects, and the expected related effects to ESA-listed species (e.g., impacts to sea turtle nesting beaches and hatchling sex ratios, associated effects to prey species, etc.) stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. In some instances, species' behavioral changes may mitigate some of the impacts, including shifting breeding season and location to avoid warmer temperatures. For example, the start of the nesting season for loggerheads has already shifted as the climate has warmed (Weishampel et al. 2004). As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the NWA DPS loggerhead sea turtle population.

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2008) was written prior to the loggerhead sea turtle DPS listings. However, this plan deals with the populations that comprise the current NWA DPS and is, therefore, the best information on recovery criteria and goals for the DPS. The plan's recovery goal for loggerhead sea turtles is "to ensure 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:

- 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.
- 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.
- 10. Minimize bycatch in domestic and international commercial and artisanal fisheries.
- 11. Minimize trophic changes from fishery harvest and habitat alteration.

The recovery plan anticipates that, with implementation of the plan, the NWA DPS 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.

Ensuring that the number of nests in each recovery unit is increasing is the recovery plans first recovery objective and, moreover, is the plan's overarching objective with 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 4,300 loggerhead sea turtles over a 10-year period 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 also does not conflict with Recovery Objectives 2 and 10. While bycatch of neritic juveniles may still occur during the proposed action, bycatch minimization measures are in place in these fisheries that avoid or minimize lethal bycatch. Further, the expansion of the TED requirements to the skimmer trawl fisheries further supports these recovery objectives. For these reasons, we do not believe the proposed action will impede achieving these recovery objectives. Likewise, we do not believe the proposed action conflicts with Recovery Objective 11, as there is no indication the shrimp fisheries analyzed in this Opinion are causing any trophic changes that would affect the NWA DPS of loggerhead sea turtles. For these reasons, we do not believe the proposed action will impede achieving this recovery objective.

The potential for 4,300 loggerhead sea turtle mortalities from the NWA DPS over the next 10 years and beyond will result in a reduction in numbers when they occur, but it is unlikely to have any detectable influence on the trends noted above, even when considered in context with information in Sections 3 (Status of the Species), 4 (Environmental Baseline), and 6 (Cumulative Effects) discussed in this Opinion. Similarly, we do not expect the nonlethal capture of 141,040 loggerhead sea turtles from the NWA DPS to have any detectable influence on the recovery objectives. Therefore, we conclude the proposed action considered in this Opinion—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NWA DPS of loggerhead sea turtles in the wild.

7.4 Leatherback Sea Turtle

We estimate that the proposed action will result in a total of 260 captures and 10 mortalities of leatherback sea turtles over the next 10 years. The nonlethal capture of 250 leatherback sea turtles (260 captures - 10 mortalities from direct effects and PIM = 250 nonlethal captures) over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The mortality of 10 leatherback sea turtles every 10 years will reduce the number of leatherback sea turtles as compared to the number that would have been present in the absence of the proposed action (assuming all other variables remained the same). These mortalities could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and otherwise survived to reproduce in the future. A leatherback sea turtle will lay multiple nests (clutches) each year. For leatherbacks, eggs per clutch is 82 for the western Atlantic, and clutch frequency averages 5.5 nests per year (NMFS and USFWS 2020).[23] Therefore, an adult female leatherback sea turtle can produces hundreds of eggs per nesting season. Although a significant portion of the eggs can be infertile (NMFS and USFWS 2020), the annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. Thus, the death of any female leatherbacks that would have otherwise survived to reproduce would eliminate the individual's and its future offspring's contribution to future generations. The anticipated lethal interactions are expected to occur anywhere in the action area. Given that these sea turtles generally have large ranges in which they disperse, no reduction in the distribution of leatherback sea turtles is expected from the proposed action.

[23] While NMFS and USFWS (2020) concluded that 7 populations met the criteria for DPSs, the species continues to be listed at the global level (85 FR 48332, August 10, 2020).

Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects, result in adverse effects on population dynamics that are appreciable. In Section 3 (Status of the Species), we reviewed the status of leatherback sea turtles in terms of nesting, female population trends, and several of the most recent assessments based on population modeling. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. Those actions include the Atlantic Pelagic Longline fishery among others, which are known to interact with the species. We also included an extensive section on Climate Change in Section 4.4. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

We believe the proposed actions are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of leatherback sea turtles in the wild. The most recent published assessment, the leatherback status review, estimated that the total index of nesting female abundance for the NWA DPS is 20,659 females (NMFS and USFWS 2020). Approximately 0.005% of the population (1 mortalities/20,659 nesting females) is anticipated to die annually through the proposed action. It should be noted that the abundance estimate is for nesting females only (i.e., does not include earlier life stages such as juveniles or adult males); therefore, the percent of the population that dies due to the proposed action is expected to be less than the percentage estimated here. Although the anticipated mortalities would result in a reduction in absolute population numbers, it is not likely this reduction would appreciably reduce the likelihood of survival of this species. If the hatchling survival rate to maturity is greater than the mortality in the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of sea turtles unaffected by the proposed action. Considering the number of lethal interactions relative to the population size, we believe the proposed action is not likely to have an appreciable effect on overall population trends.

Fisheries bycatch has been identified as a threat to leatherback sea turtles. The Leatherback Working Group noted that leatherback entanglements in vertical line fisheries (e.g., pot gear targeting crab, lobster, conch, fish) in continental shelf waters off New England, USA, and Nova Scotia, Canada, were a potential mortality sink that require continued monitoring and bycatch reduction efforts. However, the majority of documented fisheries bycatch and mortality of leatherback sea turtle has occured in fisheries outside of the shrimp fisheries considered in this Opinion. Across the range of the DPS, thousands of mature individuals are lost annually due to gillnet bycatch (especially off nesting beaches). In particular, studies estimate that well over 1,000 leatherback turtles die annually due to drift and bottom-set gillnets off Trinidad (Lum 2006; NMFS and USFWS 2020). Longline bycatch is also considered to be a widespread threat to the DPS, likely resulting in the loss of thousands of individuals annually.

As discussed in Section 4 (Environmental Baseline), although no direct leatherback impacts (i.e., oiled sea turtles or nests) from the DWH oil spill event in the northern Gulf of Mexico were observed, some impacts from that event may be expected. However, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would change the species' status to an extent that the expected interactions from these fisheries would result in a detectable change in the population status of leatherback sea turtles. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions could be seen as reducing the likelihood of survival and recovery of the species. Furthermore, we have taken regulatory actions detailed in Section 4 to reduce anthropogenic effects to Atlantic leatherbacks. For example, we have implemented measures to reduce the number and severity of leatherback interactions in the U.S. Atlantic longline fisheries. Reducing the number of leatherback sea turtles injuries and mortalities from other fisheries is expected to increase the number of Atlantic leatherbacks and increase leatherback reproduction in the Atlantic. Since most of these regulatory measures have been in place for several years now, it is likely that current nesting trends reflect the benefit of these measures to Atlantic leatherback sea turtles. There are no new known sources of mortality for leatherback sea turtles in the Atlantic other than potential impacts from the DWH oil spill event.

As mentioned in previous sections, some of the likely effects commonly associated with climate change are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The potential effects, and the expected related effects to ESA-listed species (e.g., impacts to sea turtle nesting beaches and hatchling sex ratios, associated effects to prey species, etc.) stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. Leatherbacks, however, may be more resilient to climate change in the Northwest Atlantic because of their wide geographic distribution, low nest-site fidelity, and gigantothermy (Dutton et al. 1999; Fuentes et al. 2013; Robinson et al. 2009). As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the leatherback sea turtle population

Based on the information provided above, the loss of 10 leatherback sea turtles over the next 10 years and beyond in the Atlantic due to the proposed action will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic. This is due to both the relatively large population size and the measures already taken to reduce the number of Atlantic leatherback sea turtles that are injured or die in the Atlantic. The proposed action has no effect on leatherback sea turtles that occur outside of the Atlantic. Given that the operation of the fisheries will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic, it will not appreciably reduce the likelihood of survival of the species.

The recovery plan for Atlantic leatherback sea turtles (NMFS and USFWS 1992) lists the following recovery objective, which is relevant to the proposed actions in this Opinion:

- The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico; St. Croix, U.S. Virgin Islands; and along the east coast of Florida.

We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. Since we concluded that the potential loss of leatherback sea turtles is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede 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. Therefore, we conclude the proposed action considered in this Opinion—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the leatherback sea turtle in the wild.

7.5 Hawksbill Sea Turtle

We estimate that the proposed action will result in a total of 340 captures and 10 mortalities of hawksbill sea turtles over the next 10 years. The nonlethal capture of 330 hawksbill sea turtles (340 captures - 10 mortalities from direct effects and PIM = 330 nonlethal captures) over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The mortality of 10 hawksbill sea turtles every 10 years will reduce the number of hawksbill sea turtles as compared to the number that would have been present in the absence of the proposed action (assuming all other variables remained the same). These mortalities could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and otherwise 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 any females 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. Sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of hawksbill sea turtles is expected from these takes. Likewise, as explained in Section 4 (Environmental Baseline), while a few individuals were found to have been impacted by the DWH oil spill event, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from southeast U.S. shrimp fisheries would result in a detectable change in the population status of hawksbill turtles in the Atlantic. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed action could be seen as reducing the likelihood of survival and recovery of the species.

Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects are to the extent that adverse effects on population dynamics are appreciable. In Section 3 (Status of the Species), we reviewed the status of hawksbill sea turtles in terms of nesting, female population trends, and several of the most recent assessments based on population modeling. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. We also included an extensive section on Climate Change in Section 4.4. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

We believe hawksbill sea turtles have 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. 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 abundance. The most recent 5-year status review estimated between 22,000 and 29,000 adult females existed in the Atlantic basin in 2007 (NMFS and USFWS 2013b); this estimate does not include juveniles of either sex or mature males. The potential loss of up to 10 hawksbill sea turtles every 10 years would equal only ~0.02% 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 and USFWS 2013b).

As mentioned in previous sections, some of the likely effects commonly associated with climate change are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The potential effects, and the expected related effects to ESA-listed species (e.g., impacts to sea turtle nesting beaches and hatchling sex ratios, associated effects to prey species, etc.) stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the hawksbill sea turtle population.

We have still seen positive trends in the status of this species even with the operation of the shrimp fisheries. 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 10 hawksbill sea turtles every 10 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:

- The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests at five index beaches, including Mona Island and BIRNM.
- The numbers of adults, sub-adults, and juveniles are increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, U.S. Virgin Islands, and Florida.

The recovery plan lists 6 major actions that are needed to achieve recovery, including:

- Provide long-term protection to important nesting beaches.
- Ensure at least 75% hatching success rate on major nesting beaches.
- Determine distribution and seasonal movements of turtles in all life stages in the marine environment.
- Minimize threat from illegal exploitation.
- End international trade in hawksbill products.
- Ensure long-term protection of important foraging habitats

The proposed action could cause the loss of up to 10 hawksbill sea turtles every 10 years and beyond, which may be either adult, subadult, or juvenile, and either male or female. Our evaluation of potential future mortality is based on past interactions, and even with operation of the shrimp fisheries, we have still seen positive trends in the status of this species. We determined the potential bycatch mortality 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 10 hawksbill sea turtles every 10 years will have any detectable influence over the numbers of adults, subadults, and juveniles occurring at 5 key foraging areas. Unlike the case for some other sea turtle species, none of the major actions specified for recovery are specific to shrimp fisheries bycatch or even fishery bycatch in general. In consideration of the above, we conclude the proposed action considered in this Opinion—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the hawksbill sea turtle in the wild.

7.6 Atlantic Sturgeon

The proposed action covered under this Opinion may result in 390 captures and 90 mortalities of Atlantic sturgeon over the next 10 years. Because subadult and adult Atlantic sturgeon mix extensively in the marine and estuarine environments, individuals from all 5 Atlantic sturgeon DPSs could occur within the action area. Therefore, we must determine from which DPSs the takes will occur. As described in Section 3.2.7, USGS completed a draft MSA specific to the Southeast Region (USGS unpublished data). We used the information from that report to calculate an estimate of the likely number of Atlantic sturgeon occurring the southeast (North Carolina/Virginia Border to Florida). Table 6 in Section 3.2.7 reports the likely DPS composition for individuals in the southeast as reported in the MSA (USGS unpublished data) and our estimates of the likely minimum number of individuals from each DPS occurring in the southeast; we present this information again below in Table 37.

Table 37. DPS Composition and Minimum Number of Individuals in the Southeast from Table 6 (Section 3.2.7).

Proportion of Individuals from Each DPS	Minimum Number of Individuals in Southeast by DPS
South Atlantic – 52.9% (49.9%-57.0%)	5,067
Carolina – 33.8% (29.2%-36.4%)	3,243
Chesapeake – 9.6% (7.9%-12.1%)	920
New York Bight – 3.6% (2.5%-4.8%)	343
Gulf of Maine – 1.0% (0-0.4%)	9

Table 38 shows the breakdown of bycatch and mortality by DPS over the next 10 years based on the DPS composition in the southeast. The following sections will provide the relevant and required analysis for each DPS, starting with the SA DPS (due to highest anticipated representation in the MSA) and then moving northward. These sections focus on the EEZ, as this is the only place where we anticipate any adverse effects from the federal action. We expect the effects from the sea turtle conservation regulations to be solely beneficial.

Table 38. Estimates of southeast U.S. shrimp fisheries otter trawl bycatch and mortality on each Atlantic sturgeon DPS over the next 10 years.

DPS (MSA %)	Try Net Bycatch/Mortalities	Standard Net Bycatch/Mortalities	Total Bycatch/Mortalities
Gulf of Maine DPS (1.0%)	0/0	4/0	4/0
New York Bight DPS (3.6%)	2/0	12/4	14/4
Chesapeake Bay DPS (9.6%)	6/0	32/8	38/8
Carolina DPS (33.8%)	20/0	112/30	132/30
SA DPS (52.9%)	32/0	174/48	206/48
Atlantic Sturgeon Total[1]	60/0	330/90	390/90

[1] Note that the total bycatch and mortality of each category by DPS may be different than bycatch and mortality of Atlantic sturgeon as a whole due to rounding issues.

Whether the estimated reductions in numbers and reproduction of this species (i.e., all 5 DPSs discussed in the following sections) would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects are to the extent that adverse effects on population dynamics are appreciable. In Section 3 (Status of the Species), we reviewed the status of the SA DPS of Atlantic sturgeon. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. We also included an extensive section on Climate Change in Section 4.4. Some of the likely effects to Atlantic sturgeon commonly associated with climate change are sea level rise affecting salinity levels (i.e., salt wedge) in rivers, rising temperatures exacerbating existing water quality problems with DO, and loss of access to spawning habitat due to drought conditions. The potential effects, and the expected related effects to ESA-listed species stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to any DPS of Atlantic sturgeon population. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

7.6.1 SA DPS

The proposed action may result in 206 captures and 48 mortalities of Atlantic sturgeon from the SA DPS over the next 10 years (Table 38). The nonlethal capture of any number of individuals from the DPS over a 10-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur anywhere within the South Atlantic region, but with the animals being released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential mortality of 48 Atlantic sturgeon from the SA DPS over a 10-year period would reduce the population of Atlantic sturgeon in the DPS by that amount. Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in Georgia prior to 1890. At the time of listing, only 6 spawning subpopulations were believed to have existed in the SA DPS: Combahee River, Edisto River, Savannah River, Ogeechee River, Altamaha River (including the Oconee and Ocmulgee tributaries), and Satilla River. Three of the spawning subpopulations in the SA DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. Peterson et al. (2008) estimated the number of spawning adults in the Altamaha River was 324 (95% CI: 143-667) in 2004 and 386 (95% CI: 216-787) in 2005. Bahr and Peterson (2016) estimated the Age-1 juvenile abundance in the Savannah River from 2013-2015 at 528 in 2013, 589 in 2014, and 597 in 2015. As described in Section 3.2.7, and based largely on these studies, we estimate 5,067 individuals (Table 37) from the SA DPS are likely to be in the southeast. Our GARFO estimated the ocean population of the SA DPS to be 14,911 individuals (Table 5).

We anticipate 48 mortalities from the SA DPS during a 10-year period is unlikely to change its status, as this loss represents a small percentage of the SA DPS population as a whole. Based on our estimates of 5,067 individuals from the SA DPS in the southeast, the mortality of 48 individuals from the DPS over 10 years would represent 0.47% of the population occurring in the southeast assuming no population growth over that time. Compared to the SA DPS ocean population of 14,911 estimated by our GARFO, the loss of 48 individuals would only represent 0.16% of the population assuming no population growth over that time. The best available information on the status of the SA DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017). The assessment determined the SA DPS abundance is "depleted" relative to historical levels. The assessment concluded there was not enough information available to assess the abundance of the DPS since the implementation of the 1998 fishing moratorium.

Both our estimates of the Atlantic sturgeon SA DPS in the southeast and GARFO's estimates represent only a percentage of the total DPS population, as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. It is also worth noting, however, the activities included in the proposed action have been ongoing for many years in the action area (e.g., 60+ years), so we believe these mortalities are unlikely to represent new sources of mortality for animals of the DPS. Instead, these mortalities are likely a more accurate reflection of the mortalities that have been ongoing over the last several years.

The loss of 48 individuals over a 10-year period may affect the reproductive potential of the SA DPS. We anticipate these mortalities could be of individuals from any sex or age class of the SA DPS. The South Atlantic federal shrimp fishery could result in the capture and mortality of juvenile, subadult, or adult Atlantic sturgeon. The potential loss of a sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of a male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in any particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles/subadults would be less, because they are generally more abundant than adults and are not yet sexually mature.

The mortalities associated with the proposed action is not likely to reduce distribution of the SA DPS, as mortalities occurring over a 10-year period could take place anywhere in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the expected captures and mortalities of 48 individuals from the SA DPS during a 10-year period and beyond will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect the SA DPS in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring. It will also not result in effects to the environment that would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the mortality of 48 Atlantic sturgeon from the SA DPS represent a small percentage of the total population of the DPS and these mortalities are unlikely to change the status or trends of the DPS as a whole. Furthermore, the loss of these 48 individuals are likely to only have a small effect on reproductive output, and the action will have only a minor and temporary effect on the distribution of Atlantic sturgeon from the SA DPS in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the SA DPS will survive in the wild.

A Recovery Plan for the SA DPS has not yet been developed. However, we completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017). The final listing rule (77 FR 5914; February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as ongoing threats. The severity of those threats varies by DPS.

While we consider the South Atlantic federal shrimp fishery as part of a major threat to Atlantic sturgeon (i.e., commercial fisheries), we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the SA DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the SA DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

The proposed action will result in 48 mortalities over a 10-year period and beyond leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers is small relative to the remaining population and, as a result, the impact on reproduction and future year classes will also be small enough not to affect the status of the DPS. As the proposed action occurs in federal waters of the South Atlantic, we do not expect it will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. Accordingly, we do not believe the proposed action will impede the recovery of the SA DPS by significantly exacerbating the effects of any of the other remaining major threats identified in the final listing rule. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the SA DPS.

7.6.2 Carolina DPS

The proposed action may result in 132 captures and 30 mortalities of Atlantic sturgeon from the Carolina DPS over the next 10 years (Table 38). The nonlethal capture of any number of individuals from the DPS over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur anywhere within the South Atlantic region, but released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential mortality of 30 Atlantic sturgeon from the Carolina DPS over a 10-year period would reduce the population of Atlantic sturgeon in the DPS by that amount. Historical fishery landings data indicate between 7,000 and 10,500 adult female Atlantic sturgeon were present in North Carolina prior to 1890 (Armstrong and Hightower 2002; Secor 2002). Secor (2002) estimates that 8,000 adult females were present in South Carolina during that same time frame. At the time of listing, the abundance for each river population within the DPS was estimated to have fewer than 300 spawning adults; estimated to be less than 3% of what they were historically (ASSRT 2007).

We anticipate 30 mortalities from the Carolina DPS during a 10-year period is unlikely to change its status, as this loss represents a small percentage of the Carolina DPS population as a whole. Based on our estimates of 3,243 individuals from the Carolina DPS in the southeast, the mortality of 30 individuals from the DPS over 10 years would represent 0.46% of the population segment occurring in the southeast assuming no population growth over that time. Compared to the Carolina DPS ocean population of 1,356 estimated by our GARFO, the loss of 30 individuals from the, would represent 1.11% of the population. However, we believe GARFO's estimate does not accurately reflect the likely population abundance of individuals from the Carolina DPS, given the distance from the Carolina DPS where those samples were collected. The best available information on the overall trend of the Carolina DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017). The Assessment determined the Carolina DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (67%) the Carolina DPS abundance has increased since the implementation of the 1998 fishing moratorium.

Both our estimates of the Atlantic sturgeon Carolina DPS in the southeast and GARFO's estimates represent only a percentage of the total DPS population, as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. It is also worth noting, however, the activities included in the proposed action have been ongoing for many years in the action area (e.g., 60+ years), so we believe these mortalities are unlikely to represent new sources of mortality for animals of the DPS. Instead, these mortalities are likely a more accurate reflection of the mortalities that have been ongoing over the last several years.

The loss of 30 individuals over a 10-year period may affect the reproductive potential of the Carolina DPS. We anticipate these mortalities could be of individuals from any sex or age class of the Carolina DPS population. The South Atlantic federal shrimp fishery could result in the capture and mortality of juvenile, subadult, or adult Atlantic sturgeon. The potential loss of a sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of a male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in any particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles/subadults would be less, because they are generally more abundant than adults and are not yet sexually mature.

The mortalities associated with the proposed action is not likely to reduce distribution of the Carolina DPS, as mortalities occurring over a 10-year period could take place anywhere in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the expected captures and mortalities of 30 individuals from the Carolina DPS during a 10-year period and beyond will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect the Carolina DPS in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring. It will also not result in effects to the environment that would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the mortality of 30 Atlantic sturgeon from the Carolina DPS represent a small percentage of the total population of the DPS and these mortalities are unlikely to change the status or trends of the DPS as a whole. Furthermore, the loss of these 30 individuals are likely to only have a small effect on reproductive output, and the action will have only a minor and temporary effect on the distribution of Atlantic sturgeon from the Carolina DPS in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the Carolina DPS will survive in the wild.

A Recovery Plan for the Carolina DPS has not yet been developed. However, we completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017). The final listing rule (77 FR 5914; February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as ongoing threats. The severity of those threats varies by DPS.

While we consider the South Atlantic federal shrimp fishery as part of a major threat to Atlantic sturgeon (i.e., commercial fisheries), we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the Carolina DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the Carolina DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

The proposed action will result in 30 mortalities during a 10-year period and beyond, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers is small relative to the remaining population and, as a result, the impact on reproduction and future year classes will also be small enough not to affect the status of the DPS. As the proposed action occurs in federal waters of the South Atlantic, we do not expect it will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. Accordingly, we do not believe the proposed action will impede the recovery of the Carolina DPS by significantly exacerbating the effects of any of the other remaining major threats identified in the final listing rule. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the Carolina DPS.

7.6.3 Chesapeake Bay DPS

The proposed action may result in 38 captures and 8 mortalities of Atlantic sturgeon from the Chesapeake Bay DPS over the next 10 years (Table 38). The nonlethal capture of any number of individuals from the DPS over 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur anywhere within the South Atlantic region, but released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential mortality of 8 Atlantic sturgeon from the Chesapeake Bay DPS over a 10-year period would reduce the population of Atlantic sturgeon in the DPS by that amount. Historically, the Chesapeake Bay DPS likely supported more than 10,000 spawning adults

(ASSRT 2007; Maine State Planning Office 1993; Secor 2002). Currently, there are 4 known spawning subpopulations for the Chesapeake Bay DPS, one each for the Pamunkey River and for Marshyhope Creek, and 2 for the James River (Balazik et al. 2017; Balazik et al. 2012a; Balazik and Musick 2015; Greenlee et al. 2017; Hager et al. 2014; Richardson and Secor 2017).

We anticipate 8 mortalities from the Chesapeake Bay DPS over the next 10 years is unlikely to change its status, as this loss represents a small percentage of the Chesapeake Bay DPS population as a whole. As noted in Section 3.2.7, our current understanding of the migratory behavior of Atlantic sturgeon suggests these animals would be transient individuals that are unlikely to represent a significant portion of the total population from the DPS. Based on our estimates of 920 individuals from Chesapeake Bay DPS in the southeast (Table 37), the mortality of 8 individuals from the DPS over 10 years would represent 0.43% of the population segment occurring in the southeast assuming no population growth over that time. Compared to the Chesapeake Bay DPS ocean population of 8,811 estimated by our GARFO, however, the loss of 8 individuals, would only represents 0.04% of the population. We believe GARFO's estimate is based on information that more accurately reflects the likely population abundance of individuals from this DPS. The best available information on the overall trend of the Chesapeake Bay DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017). The Assessment determined the Chesapeake Bay DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (37%) the Chesapeake Bay DPS abundance has increased since the implementation of the 1998 fishing moratorium.

Both our estimates of the Atlantic sturgeon Chesapeake Bay DPS in the southeast and GARFO's estimates represent only a percentage of the total DPS population, as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. It is also worth noting, however, the activities included in the proposed action have been ongoing for many years in the action area (e.g., 60+ years), so we believe these mortalities are unlikely to represent new sources of mortality for animals of the DPS. Instead, these mortalities are likely a more accurate reflection of the mortalities that have been ongoing over the last several years.

The loss of 8 individuals over a 10-year period may affect the reproductive potential of the Chesapeake Bay DPS. We anticipate these mortalities could be of individuals from any sex or age class of the Chesapeake Bay DPS population. The South Atlantic federal shrimp fishery could result in the capture and mortality of juvenile, subadult, or adult Atlantic sturgeon. The potential loss of a sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of a male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in any particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles/subadults would be less, because they are generally more abundant than adults and are not yet sexually mature.

The mortalities associated with the proposed action is not likely to reduce distribution of the Chesapeake Bay DPS, as mortalities occurring over a 10-year period could take place anywhere in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the expected captures and mortalities of 8 individuals from the Chesapeake Bay DPS during a 10-year period will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect the Chesapeake Bay DPS in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring. It will also not result in effects to the environment that would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the mortality of 8 Atlantic sturgeon from the Chesapeake Bay DPS represent a small percentage of the total population of the DPS and these mortalities are unlikely to change the status or trends of the DPS as a whole. Furthermore, the loss of these 8 individuals are likely to only have a small effect on reproductive output, and the action will have only a minor and temporary effect on the distribution of Atlantic sturgeon from the Chesapeake Bay DPS in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the Chesapeake Bay DPS will survive in the wild.

A Recovery Plan for the Chesapeake Bay DPS has not yet been developed. However, we completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017). The final listing rule (77 FR 5914; February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as ongoing threats. The severity of those threats varies by DPS.

While we consider the South Atlantic federal shrimp fishery as part of a major threat to Atlantic sturgeon (i.e., commercial fisheries), we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the Chesapeake Bay DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the Chesapeake Bay DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

The proposed action will result in 8 mortalities during a 10-year period and beyond, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers is small relative to the remaining population and, as a result, the impact on reproduction and future year classes will also be small enough not to affect the status of the DPS. And as the proposed action occurs in federal waters of the South Atlantic, we do not expect it will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. Accordingly, we do not believe the proposed action will impede the recovery of the Chesapeake Bay DPS by significantly exacerbating the effects of any of the other remaining major threats identified in the final listing rule. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the Chesapeake Bay DPS.

7.6.4 New York Bight DPS

The proposed action may result in 14 captures and 4 mortalities of Atlantic sturgeon from the New York Bight DPS during a 10-year period (Table 38). The nonlethal capture of any number of individuals from the DPS over a 10-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur anywhere within the South Atlantic region, but released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential mortality of 4 Atlantic sturgeon from the New York Bight DPS over a 10-year period would reduce the population of Atlantic sturgeon in the DPS by that amount. Prior to the onset of expanded fisheries exploitation of sturgeon in the 1800s, a conservative historical estimate for the Hudson River Atlantic sturgeon population was 10,000 adult females (Secor 2002). Current population abundance is likely at least one order of magnitude smaller than historical levels (ASSRT 2007; Kahnle et al. 2007; Secor 2002). Based on data collected from 1985-1995, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population (Kahnle et al. 2007). There is no abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800s indicate that this was historically a large population, with an estimated 180,000 adult females prior to 1890 (Secor 2002; Secor and Waldman 1999). Based on the capture of juvenile Atlantic sturgeon in the Delaware River, researchers estimated estimate there were 3,656 (95% CI = 1,935–33,041) Age 0-1 juvenile Atlantic sturgeon in the Delaware River subpopulation in 2014 (Hale et al. 2016). However, the relatively low numbers of captured adults suggest the existing riverine subpopulation is limited in size. For example, of the 261 adult-sized Atlantic sturgeon captured for scientific purposes off the Delaware Coast between

2009 and 2012, 100 were subsequently identified by genetics analysis to belong to the Hudson River subpopulation while only 36 belonged to the Delaware River subpopulation (Wirgin et al. 2015). The Atlantic Sturgeon Status Review Team (ASSRT 2007) suggested there may be less than 300 spawning adults per year for the Delaware River portion of the New York Bight DPS.

We estimate 4 mortalities from the New York Bight DPS during a 10-year period is unlikely to change its status, as this loss represents a small percentage of the New York Bight DPS population as a whole. As noted in Section 3.2.7, our current understanding of the migratory behavior of Atlantic sturgeon suggests these animals would be transient individuals that are unlikely to represent a significant portion of the total population from the DPS. Based on our estimates of 343 individuals from New York Bight DPS in the southeast (Table 37), the mortality of 4 individuals from the DPS over 10 years would represent 0.58% of the population segment occurring in the southeast assuming no population growth over that time. Compared to the New York Bight DPS ocean population of 34,566 estimated by our GARFO, the loss of 4 individuals would only represents 0.006% of the population. We believe GARFO's estimate is based on information that more accurately reflects the likely population abundance of individuals from this DPS. The best available information on the overall trend of the New York Bight DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017). This Assessment determined the New York Bight DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (75%) the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium.

Both our estimates of the Atlantic sturgeon New York Bight DPS in the southeast and GARFO's estimates represent only a percentage of the total DPS population, as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. It is also worth noting, however, the activities included in the proposed action have been ongoing for many years in the action area (e.g., 60+ years), so we believe these mortalities are unlikely to represent new sources of mortality for animals of the DPS. Instead, these mortalities are likely a more accurate reflection of the mortalities that have been ongoing over the last several years.

The loss of 4 individuals over a 10-year period may affect the reproductive potential of the New York Bight DPS. We anticipate these mortalities could be of individuals from any sex or age class of the New York Bight DPS population. The South Atlantic federal shrimp fishery could result in the capture and mortality of juvenile, subadult, or adult Atlantic sturgeon. The potential loss of a sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of a male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in any particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles/subadults would be less, because they are generally more abundant than adults and are not yet sexually mature.

The mortalities associated with the proposed action is not likely to reduce distribution of the New York Bight DPS, as mortalities occurring over a 10-year period could take place anywhere in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the expected captures and mortalities of 4 individuals from the New York Bight DPS during a 10-year period and beyond will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect the New York Bight DPS in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring. It will also not result in effects to the environment that would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the mortality of 4 Atlantic sturgeon from the New York Bight DPS represent a small percentage of the total population of the DPS and these mortalities are unlikely to change the status or trends of the DPS as a whole. Furthermore, the loss of these 4 individuals are likely to only have a small effect on reproductive output, and the action will have only a minor and temporary effect on the distribution of Atlantic sturgeon from the New York Bight DPS in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the New York Bight DPS will survive in the wild.

A Recovery Plan for the New York Bight DPS has not yet been developed. However, we completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017). The final listing rule (77 FR 5914; February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as ongoing threats. The severity of those threats varies by DPS.

While we consider the South Atlantic federal shrimp fishery as part of a major threat to Atlantic sturgeon (i.e., commercial fisheries), we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the New York Bight DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the New York Bight DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

The proposed action will result in 4 mortalities during a 10-year period and beyond, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers is small relative to the remaining population and, as a result, the impact on reproduction and future year classes will also be small enough not to affect the status of the DPS. And as the proposed action occurs in federal waters of the South Atlantic, we do not expect it will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. Accordingly, we do not believe the proposed action will impede the recovery of the New York Bight DPS by significantly exacerbating the effects of any of the other remaining major threats identified in the final listing rule. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the New York Bight DPS.

7.6.5 Gulf of Maine DPS

As we do not estimate any mortalities of Atlantic sturgeon from the Gulf of Maine DPS as a result of the proposed action, we expect no associated reduction in the reproduction, numbers, or distribution of Atlantic sturgeon from this DPS. We likewise do not expect the proposed action to cause an appreciable reduction in the likelihood of both the survival and the recovery of the Gulf of Maine DPS.

7.6.6 Atlantic Sturgeon Summary

The proposed action considered in this Opinion is expected to result in the incidental bycatch of 390 Atlantic sturgeon and anticipated mortalities of 90 Atlantic sturgeon over a 10-year period and beyond. We conclude these effects—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of any of the 5 Atlantic sturgeon DPSs in the wild.

7.7 Gulf Sturgeon

The proposed action covered under this Opinion may result in 10 captures and 2 mortalities of Gulf sturgeon during a 10-year period. The nonlethal capture of 8 Gulf sturgeon (10 captures - 2 mortalities from direct effects and PIM = 8 nonlethal captures) every 10 years is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Nonlethal captures will not result in a reduction in numbers of the species, as we anticipate these nonlethal captures to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur anywhere within the action area and would be released within the same general area where caught, we anticipate no change in the distribution of Gulf sturgeon. While the mortality of 2 Gulf sturgeon will reduce the overall population and potential reproductive output, the reduction is not likely to appreciably reduce the likelihood of survival for Gulf sturgeon. The number of individuals within each riverine populations is variable across their range, but generally over the last decade (USFWS and NMFS 2009) populations in the eastern part of the range (Suwannee, Apalachicola, Choctawhatchee) appear to be relatively stable in number or have a slightly increasing population trend. The action will not affect Gulf sturgeon in a way that prevents the species from having a sufficient population, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Gulf sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter (i.e., it will not increase the risk of extinction faced by this species). The loss of only 2 individuals over a 10-year period and beyond will not significantly decrease the overall population of Gulf sturgeon or reduce its distribution. Additionally, the proposed action will not create any barrier to pre-spawning sturgeon accessing the overwintering sites or impede Gulf sturgeon from accessing any seasonal concentration areas, including foraging, spawning or overwintering grounds in the Gulf of Mexico.

Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects are to the extent that adverse effects on population dynamics are appreciable. In Section 3 (Status of the Species), we reviewed the status of Gulf sturgeon. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. We also included an extensive section on Climate Change in Section 4.4. Similar to the potential climate change related effects facing Atlantic sturgeon, warmer water, sea level rise, and higher salinity levels could lead to accelerated changes in habitats utilized by Gulf sturgeon. Higher water temperatures combined with increased nutrients from storm runoff due to climate change may also result in increased invasive submerged and emergent water plants and phytoplankton, which are the foundation of the food chain (FWC 2009). The potential effects, and the expected related effects to ESA-listed species stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the Gulf sturgeon population. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

Recovery is defined as the improvement in status such that listing is no longer appropriate. The Gulf Sturgeon Recovery/Management Plan was created in 1995 (USFWS and GSMFC 1995). During the most recent 5-year review (USFWS and NMFS 2009), it was determined that the 1995 criteria do not directly address the 5 statutory listing/recovery factors. Five-factor-based criteria are necessary for measuring progress towards reducing threats and for determining when the protections of the ESA are no longer necessary for the taxon. New criteria in a revised recovery plan should use demographic parameters that can be estimated from mark-recapture studies, including population abundance, and other appropriate metrics organized according to the statutory five factors. To evaluate whether the reductions in numbers and reproduction from the proposed action will appreciably reduce the Gulf sturgeons likelihood of recovery in the wild, we evaluated whether these reductions would in turn reduce the likelihood that the status of the Gulf sturgeon can improve to the point where it is recovered and could be delisted.

The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in only a small reduction in the number (i.e., 2 fish every 10 years and beyond) of Gulf sturgeon in the Gulf of Mexico and therefore, it will not affect the overall distribution of Gulf sturgeon. The reduction in numbers and future reproduction is very small and, therefore, will not change the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will cause the mortality of a small percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. We therefore conclude that the proposed action is not expected to appreciably reduce the likelihood of the Gulf sturgeon's recovery in the Gulf of Mexico.

Because Gulf sturgeon are a demersal species predominantly occurring in state waters, we do not expect other federal fisheries (e.g., other bottom trawl fisheries) to interact with this species. We discussed the potential effects of climate change on the species in Section 4.4, and noted the expected small increase in temperature and its associated impacts over the next 10 years is unlikely to cause a significant effect to Gulf sturgeon. Therefore, in summary, we believe that the effects associated with the proposed action—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of Gulf sturgeon survival or recovery in the wild.

7.8 Giant Manta Ray

We estimate that the proposed action will result in a total of 16,780 captures of giant manta ray over the next 10 years. The nonlethal capture of 16,780 giant manta ray over 10 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 throughout the action area and would be released within the general area where caught, no change in the distribution of this species is anticipated. Therefore, we believe the nonlethal take on average of 1,678 giant manta rays per year will not result in population level impacts nor will it change their distribution.

Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects are to the extent that adverse effects on population dynamics are appreciable. In Section 3 (Status of the Species), we reviewed the status of giant manta ray. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. We also included an extensive section on Climate Change in Section 4.4. Because the giant manta ray is 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). Climate change, however, may alter zooplankton abundance and distribution as a result of ocean acidification. Regardless, the potential effects, and the expected related effects to ESA-listed species stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the giant manta ray population. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

There are no current and accurate abundance estimates available, as the species tends to be only sporadically observed. For instance, there is available abundance data represented by records of over 500 individuals observed off the east coast of Florida (Miller and Klimovich 2017), but it is unclear what proportion of the global population that represents. Moreover, it is unclear if giant manta ray are found in similar densities throughout the remainder of the action area. There is no population growth rate available for the giant manta ray, however, the best available data indicate that the species has suffered population declines of significant magnitude (up to 95% in some places) in the Indo-Pacific and Eastern Pacific portion of its range. As described in Section

3, however, it is unlikely that overutilization as a result of bycatch mortality is a significant threat to giant manta rays in the Atlantic Ocean (83 FR 2916; January 22, 2018). Yet, information is severely lacking on both population sizes and distribution of the giant manta ray, as well as current catch and fishing effort on the species throughout this portion of its range. The species is not considered to be at high risk 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 could cause drastic declines in the population.

Due to the lack of any global or large-scale regional population estimates, we are compelled to make an informed judgement by applying the available, albeit limited, information to evaluate if the effects of the proposed action may appreciably reduce the likelihood of giant manta ray surviving in the wild. Studies indicate local to regional subpopulations range widely from 100-1,875 individuals (Miller and Klimovich 2017; Beale et al. 2019). Some of these studies rely on diver observations from a very limited number of sites to estimate local and regional subpopulations. Given the South Atlantic (i.e., North Carolina south to the Dry Tortugas, Florida) and the Gulf of Mexico regions are considerably larger than the areas considered in available literature, we will utilize the high end of the range (i.e., 1,875) for our population estimates for each area, given the lack of other available information. Nonetheless, we conclude the nonlethal take of 1,678 giant manta rays per year will not result in population level impacts, nor will it change the species' distribution.

As described in Sections 4 and 4.4, effects from U.S. fishing have resulted in interactions with giant manta rays and large-scale impacts that affect ocean temperatures, currents, and potentially food chain dynamics, may pose a threat to this species. However, given the migratory behavior of the giant manta ray and tolerance to both tropical and temperate waters, these animals likely have the ability to shift their range or distribution to remain in an environment conducive to their physiological and ecological needs, providing the species with resilience to these effects.

Since giant manta rays were recently listed, a recovery plan for them is not yet available. The first step in recovering a species, however, is to reduce identified threats; only by alleviating these threats can we achieve lasting recovery. The Final Listing Rule (83 FR 2916, January 22, 2018) noted that overall, current management measures that are in place for fishers 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, we do 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). 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 is not likely to impede giant manta rays from continuing to survive and will not impede the process of restoring the ecosystems that affect giant manta rays. The proposed action will not have any detectable effect on the overall size of the population; we do not expect it to affect the giant manta ray's ability to meet its lifecycle requirements and to retain the potential for recovery; and operation of the fisheries will not alter the rates of dispersal and gene flow. Based on the evidence available, we conclude the estimated nonlethal bycatch of 16,780 giant manta rays every 10 years and beyond associated with the effects of the proposed action—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to appreciably reduce the likelihood of giant manta ray surviving and recovering in the wild.

7.9 Smalltooth Sawfish

We estimate that the proposed action will result in a total of 3,612 captures and 1,806 mortalities of smalltooth sawfish over the next 10 years. The nonlethal capture of 1,806 smalltooth sawfish (3,612 captures - 1,806 mortalities from direct effects and PIM = 1,806 nonlethal captures) over 10 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 throughout the action area and would be released within the general area where caught, no change in the distribution of this species is anticipated. Therefore, we believe the nonlethal take on average of 181 smalltooth sawfish per year (1,806 / 10 = 180.6 per year) will not result in population level impacts nor will it change their distribution. The mortality of 1,806 smalltooth sawfish over a 10-year period will reduce the number of smalltooth sawfish as compared to the number that would have been present in the absence of the proposed action (assuming all other variables remained the same). These mortalities could also result in a potential reduction in future reproduction, assuming some proportion of these individuals would be female and otherwise survived to reproduce in the future. An adult female smalltooth sawfish may have a litter of approximately 10 pups probably every 2 years, and because smalltooth sawfish produce more well-developed young, it is likely that some portion of these pups would have survived. Thus, the death of any females eliminates any individual's contribution to future generations, and the proposed action would result in a reduction in future smalltooth sawfish reproduction. A reduction in the distribution of the smalltooth sawfish is not expected as the anticipated lethal interactions are expected to be dispersed throughout the range of smalltooth sawfish that overlaps with the proposed action (i.e., primarily off Florida and the Florida Keys). Whether these 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. Likewise, determining if the reductions in numbers and reproduction of this species attributed to the proposed action would appreciably reduce the species' likelihood of recovering depends on the probable effect the changes in numbers and reproduction would have on the population's growth rate, and whether the affected growth rate would allow the species to recover.

Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. That is, will the estimated reductions, when viewed within the context of the Status of the Species, Environmental Baseline, and Cumulative Effects are to the extent that adverse effects on population dynamics are appreciable. In Section 3 (Status of the Species), we reviewed the status of smalltooth sawfish. In Section 4 (Environmental Baseline), we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this species. We also included an extensive section on Climate Change in Section 4.4, which indicated potential sea level rise stemming from climate change could negatively affect mangrove habitat utilized by smalltooth sawfish. Rising water temperature may also push the species north. Regardless, the potential effects, and the expected related effects to ESA-listed species stemming from climate change are the result of a slow and steady shift over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As previously discussed, we have elected to view the effects of climate change on affected species over a more manageable and predictable 10-year time period due to this reality. And within this 10-year time period, we do not expect the effects of climate change will present a risk to the U.S. DPS smalltooth sawfish population. Section 6 (Cumulative Effects) discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area. It is important to note that virtually all of the effects already discussed, including the effects from the shrimp fisheries, have been occurring and affecting the species for decades. All of the previously discussed effects are part of the baseline upon which this analysis is founded, and the associated population level implications for the species are reflected in the species current population trends.

While the mortality of 1,806 smalltooth sawfish over 10 years will result in an instantaneous reduction in absolute population numbers, we believe the mortalities associated with the proposed action are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the U.S. DPS population of smalltooth sawfish in the wild. This is because we do not believe these mortalities will have any measurable effect on the species' population trends. The mortality of 1,806 sub-adult/adult animals is significant for a population that is currently estimated to be at a level less than 5% of its size at the time of the European settlement. However, available data summarized in Section 3, we believe available information demonstrates the smalltooth sawfish population is increasing and undergoing recovery. Furthermore, as noted in the beginning of Section 7, we are including a 5% increase to our mortality estimates for smalltooth sawfish to account for this anticipated population growth into the future. As a result, we believe the mortality of 1,806 sub-adult/adult males or females over 10 years is not expected to have any measureable impact on this population growth and recovery. This is because effort and associated smalltooth sawfish mortality in the federal shrimp fishery has decreased significantly from the amount that existed when the population doubling times of 10.3 to 13.5 years was calculated (Simpfendorfer 2000). Therefore, we believe the effects of the proposed action will not cause an appreciable reduction in the likelihood of the survival of the U.S. DPS of smalltooth sawfish in the wild.

The following analysis considers the effects of the take on the likelihood of recovery in the wild. The Smalltooth Sawfish Recovery Plan (NMFS 2009b) identifies 2 relevant recovery objectives over a period of 100 years:

- 1. Minimize human interactions and associated injury and mortality.
- 3. Ensure smalltooth sawfish abundance increases substantially and the species reoccupies areas from which it had been previously extirpated.

With full implementation of all recovery objectives, the Recovery Plan anticipates the U.S. DPS of smalltooth sawfish will recover within 100 years. The Recovery Plan includes multiple recovery actions that are particularly relevant to the proposed action of this Opinion:

- Monitor the take and fate of the species in commercial and recreational fisheries throughout the species' range.
- Improve the capacity and geographic coverage of the sawfish encounter data collection program to enable full investigation, review, and evaluation of each report of smalltooth sawfish fishery interactions.
- Determine the post-release mortality of smalltooth sawfish from various types of fishing gear.
- Integrate collection of data on smalltooth sawfish into current commercial fishery observer programs and implement new programs where required.
- Implement and adequately fund observer programs over the long term.
- Use PVA or other types of population models to evaluate the effect of fishery takes on the species' viability.
- Implement strategies to reduce bycatch, mortality, and injury, in specific fisheries to ensure the species' viability.
- Monitor trawl fisheries to ensure they do not threaten the viability of the population.
- Investigate fishing devices, gear modifications, and techniques (physical, electronic, chemical, net configuration, etc.) that reduce the likelihood of sawfish capture, improve the chances of sawfish escapement, minimize harm to sawfish and humans from capture, and facilitate successful release of healthy sawfish.
- Recommend the use of fishing devices, gear modifications, and/or techniques found to be effective at reducing bycatch of smalltooth sawfish and/or mitigating the effects of capture in areas frequented by sawfish, other important sawfish habitats, and in trawl fisheries encountering significant numbers of sawfish.

- Develop, distribute, and implement Safe Handling and Release Guidelines for smalltooth sawfish for recreational and commercial fisheries to minimize interactions, injury, and mortality.
- Investigate short-term movement patterns of adult sawfish to provide information on habitat use patterns.
- Investigate seasonal patterns of occurrence and habitat use of adults.
- Monitor abundance of adult smalltooth sawfish in aggregation areas.
- Evaluate fishery observer programs to determine their suitability to act as surveys of relative abundance of adult smalltooth sawfish.
- Analyze annual relative abundance data for adult smalltooth sawfish and determine if it meets the criteria in Objective 3.
- Conduct tagging studies, potentially using satellite and/or archival technology, to study seasonal migrations along the U.S. East Coast and within the Gulf of Mexico.
- Continue existing effective sawfish encounter reporting systems with outreach efforts throughout the historic range, with special efforts focused on the north central Gulf of Mexico, Georgia, South Carolina, and North Carolina.

We are currently funding several actions identified in the Recovery Plan for smalltooth sawfish; adult satellite tagging studies, the International Sawfish Encounter Database, and monitoring take in commercial fisheries. Additionally, we have developed Safe Handling and Release Guidelines for recreational fisheries and Sawfish Handling, Release, and Reporting Procedures for commercial fisheries (Appendix 2). Despite the ongoing threats from the Federal shrimp fisheries, we have still seen an improving trend in the status of this species.

Based on the evidence available, we conclude the estimated bycatch mortality of 1,806 smalltooth sawfish over 10 years and beyond associated with the effects of the proposed action—even amidst other ongoing threats to the species including bycatch mortality from other fisheries (Appendix 1), other federal actions (i.e., anticipated take issued in other Opinions), and/or and the potential effects of climate change—is not expected to appreciably reduce the likelihood of smalltooth sawfish surviving and recovering in the wild.

8 INCIDENTAL TAKE STATEMENT

Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the RPMs and the terms and conditions of the ITS of the Opinion. Take that occurs while fishing not in compliance with the requirements of the proposed action does not constitute authorized incidental take because it is not incidental to an otherwise lawful activity. Accordingly, such take is not covered by the ITS and constitutes unlawful take.

Section 7(b)(4)(c) of the ESA specifies that to provide an ITS for an endangered or threatened species of marine mammal, the taking must be authorized under Section 101(a)(5) of the MMPA. Since no incidental take of listed marine mammals is expected or has been authorized under Section 101(a)(5) of the MMPA, no statement on incidental take of protected marine mammals is provided and no take is authorized. F/SER2 must immediately notify our Office of Protected Resources should a take of a listed marine mammal occur.

This Opinion establishes an ITS with RPMs and terms and conditions for incidental take coverage for sea turtle takes throughout the action area, and for Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish takes in the federal shrimp fishery. We have not issued an ESA Section 4(d) rule prohibiting the take of Gulf sturgeon or giant manta ray so no incidental take coverage is needed, despite expected takes in the federal fishery. Consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012), however, our ITS is included to serve as a check on the no-jeopardy conclusion by providing a reinitiation trigger if the level of take analyzed in the Opinion is exceeded.

8.1 Anticipated Incidental Take

The level of takes occurring annually is variable and influenced by sea temperatures, species abundances, fishing effort, and other factors that are difficult to predict. The significant events of 2020 are a good example, as both the impacts of COVID-19 and the most active Atlantic hurricane season in history have had significant impacts on the shrimp fisheries (e.g., effort and landings). Because of this variability, it is unlikely that all species evaluated in this Opinion will be consistently impacted year after year. For example, some years may have no observed or otherwise documented interactions and, thus, no estimated take will occur. As a result, monitoring fisheries using 1-year estimated take levels is largely impractical. Additionally, given the rarity of observed captures in the shrimp fisheries of sea turtles and other data limitations, as well as the effort required to produce comparable Bayesian bycatch model results (i.e., Babcock et al. (2018)), as well as the review of observer data and application of PIM, producing annual estimates is not practical. It normally takes over a year to process, analyze, and peer-review data once available for a valid bycatch estimate. With current resources and other agency demands, it is neither reasonable nor possible to estimate bycatch annually. Furthermore, annual estimates are unlikely to change considerably such that they affect our jeopardy analyses. Warden et al. (2015) state "when the population is large compared to the incidental mortality, frequent (e.g., annual) monitoring is not likely to produce results that are substantially different from the previous assessment. Less frequent but more comprehensive assessments, which explicitly address uncertainty, may provide more reliable information." For these reasons, and based on our experience monitoring fisheries, we believe a 5-year time period is appropriate for meaningful monitoring of take with respect to the ITS. Thus, take estimates will be provided based on 5-year intervals for the life of this Opinion. As this Opinion provides the first complete analyses of total anticipated bycatch for sea turtles (i.e., direct observed bycatch estimated in Babcock et al. (2018) combined with anticipated PIM), the issuance of this Opinion would start the clock for the next 5-year period; new bycatch estimates for all affected species will be required by or before the end of the next 5-year period. Tables 39-42 display the anticipated take of listed species encompassed by this Opinion over the 5-year monitoring periods.

Table 39. Incidental otter trawl takes in the southeast U.S. shrimp fisheries anticipated over the 5-year monitoring periods. Rows highlighted in red indicate species (i.e., green sea turtle and smalltooth sawfish) with adjusted capture and mortality estimates that take into consideration anticipated population growth.

Species	Try Nets		Standard Nets		Total Mortalities
	Captures	Mortalities	Captures	Mortalities	
Kemp's Ridley Sea Turtle	21,060	385	29,005	2,200	2,585
Loggerhead Sea Turtle	53,015	1,010	16,525	600	1,610
Green Sea Turtle	5,891	194	13,090	1,119	1,313
Leatherback Sea Turtle	25	0	105	5	5
Hawksbill Sea Turtle	30	0	140	5	5
Atlantic Sturgeon	30	0	165	45	45
Gulf Sturgeon	-	-	5	1	1
Smalltooth Sawfish	-	-	1,806	903	903
Giant Manta Ray	-	-	8,390	0	0

Table 40. Incidental take of Atlantic sturgeon by DPS in the southeast U.S. shrimp fisheries anticipated over the 5-year monitoring periods.

DPS (MSA %)	Try Net Bycatch/Mortalities	Standard Net Bycatch/Mortalities	Total Bycatch/Mortalities
Gulf of Maine DPS (1.0%)	0/0	2/0	2/0
New York Bight DPS (3.6%)	1/0	6/2	7/2
Chesapeake Bay DPS (9.6%)	3/0	16/4	19/4
Carolina DPS (33.8%)	10/0	56/15	66/15
SA DPS (52.9%)	16/0	87/24	103/24
Atlantic Sturgeon Total[1]	30/0	165/45	195/45

[1] Note that the total bycatch and mortality of each category by DPS may be different than bycatch and mortality of Atlantic sturgeon as a whole due to rounding issues.

Table 41. Incidental skimmer trawl takes in the southeast U.S. shrimp fisheries anticipated over the 5-year monitoring periods. Green sea turtle takes highlighted in red are adjusted to take into consideration anticipated population growth.

Species	Skimmer Trawl Captures	Skimmer Trawl Mortalities
Kemp's Ridley Sea Turtle	34,430	5,920
Loggerhead Sea Turtle	3,130	540
Green Sea Turtle	2,233	387

Table 42. Total (otter and skimmer trawl fisheries, all nets combined) incidental sea turtle takes in the southeast U.S. shrimp fisheries anticipated over the 5-year monitoring periods. Green sea turtle takes highlighted in red are adjusted to take into consideration anticipated population growth.

Species	Captures	Mortalities
Kemp's Ridley Sea Turtle	84,495	8,505
Loggerhead Sea Turtle	72,670	2,150
Green Sea Turtle	21,214	1,700
Leatherback Sea Turtle	130	5
Hawksbill Sea Turtle	170	5

8.2 Effect(s) of the Take

We have determined that the anticipated take specified in Section 8.1 is not likely to jeopardize the continued existence of Kemp's ridley, green (NA and SA DPSs), loggerhead (NWA DPS), leatherback, and hawksbill sea turtles, as well as Atlantic sturgeon (all 5 DPSs), Gulf sturgeon, giant manta ray, and smalltooth sawfish (U.S. DPS) as a result of the proposed action.

8.3 Reasonable and Prudent Measures (RPMs)

Section 7(b)(4) of the ESA requires us 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 document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. These measures and terms and conditions are non-discretionary, and must be implemented for the protection of Section 7(o)(2) to apply.

We have determined that the following RPMs are necessary and appropriate to minimize impacts of the incidental take of ESA-listed species related to the proposed action. The following RPMs and associated terms and conditions are established to implement these measures, and to document incidental takes. Only incidental takes that occur while these measures are in full implementation are authorized. These restrictions remain valid until reinitiation and conclusion of any subsequent Section 7 consultation.

RPM 1: Monitoring

F/SER2 must ensure that future fisheries effort monitoring is conducted at equivalent (or greater) levels as conducted over the past 10 years. F/SER2, F/SER3, and SEFSC must use this information to complete new bycatch estimates for affected species at a prescribed interval. Fishery observers must ensure reporting of any sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish encountered in gear used by the southeast U.S. shrimp fisheries detects any adverse effects such as serious injury or mortality and includes the collection of necessary biological and life history data from individual encounters (e.g., species ID, date, location, size measurements, genetic information, photos/video, etc.). Furthermore, fishery observers must follow protocols to evaluate the condition of all sea turtles (e.g., reflex test) brought on board a shrimp trawler to allow for PIM analysis. This information is essential for conducting our effects analysis as required by the ESA.

RPM 2: Sampling

Fishery observers must collect, record, preserve, and submit all blood and genetic samples per established protocols or as discussed in Appendix 2. Likewise, SEFSC must ensure tagging of any captured sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish follows established protocols. Collecting this basic biological and genetic data on captured sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish can yield important information on fishery impacts, survivability, population dynamics, species identification, and other information essential for evaluating individual and population-level effects.

RPM 3: Ecological Studies

F/SER2 and SEFSC must analyze available data to determine if there are potential "hot spots" where elevated bycatch and bycatch-associated mortality would warrant consideration of additional protective measures (e.g., time/area closures). RPM 3 and the accompanying Terms and Conditions specify the importance of using current data already available to reduce the incidental bycatch and increase survivability of sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish. Temporal and spatial data can provide insight on where these interactions are most likely to occur, and can be paired with modifications to fishing practices to minimize the respective incidental capture and mortality of these species.

RPM 4: Handling

F/SER2 must ensure fishers (e.g., via outreach and education efforts) and observers handle sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish in a manner that prevents injury and helps ensure survivability upon release. Any captured sea turtle in a comatose or lethargic state must be retained on board, handled, resuscitated, and released according to our established procedures (see Appendix 2), as deemed practicable and in consideration of best practices for safe vessel and fishing operations. Likewise, captured Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish must be released in a manner that avoids further injury, to the maximum extent practicable (see Appendix 2). Proper handling of any protected species incidentally caught during fishery operations is essential to increase the likelihood of its survival.

8.4 Terms and Conditions

To be exempt from take prohibitions established by Section 9 of the ESA, F/SER2 must comply with the following terms and conditions, which implement the RPMs described above. These terms and conditions are mandatory.

The following terms and conditions implement RPM 1:

We must continue to monitor the southeast U.S. shrimp fisheries in order to document and report incidental bycatch and entanglement of sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish. The SEFSC must provide an update on bycatch estimates for these species within 5 years of the issuance of this Opinion; new bycatch estimates for all affected species will be required by or before the end of that time. We will continue to use records from our observer program as the primary means of collecting incidental take information. For sea turtles, the take estimates described in this Opinion were generated using a statistical model that is not feasible to conduct on an annual basis due to the data needs; length of time to develop, review, and finalize the estimates; and methodology. Given the available observer effort and general rarity of encounters with listed species, we often need to pool data across years to have enough data to produce a robust, model-based estimate of total interactions. This is reinforced by the giant manta ray bycatch estimates we include in this Opinion, which is based on only 1 year of data, and for which we noted are both likely highly uncertain and overestimate actual bycatch. Furthermore, annual estimates are unlikely to change considerably such that they affect population assessments. Warden et al. (2015) notes, "when the population is large compared to the incidental mortality, frequent (e.g., annual) monitoring is not likely to produce results that are substantially different from the previous assessment. Less frequent but more comprehensive assessments, which explicitly address uncertainty, may provide more reliable information." For these reasons, we believe monitoring and issuing take estimates over a 5-year period is reasonable and prudent.

Within 1 year of issuance of this Opinion, the observer program's data forms must be updated to include information options for giant manta ray.

The following terms and conditions implement RPM 2:

For giant manta ray:

- Do not bring giant manta rays aboard; ultimately release the animal from the net while it remains in the water.
- Estimate size; size should be recorded as disc width (i.e., straight-line measurement from wing tip to wing tip).
- Take photographs and/or video, including the handing and release of each giant manta ray.
- Observers must take tissue samples. While the animal is in the water, use a tissue-sampling pole with an attached sampling tip to remove a small amount of tissue from the dorsal surface of a wing. Store the sample in a collection tube with ethanol (if a collection tube is unavailable, store in a plastic bag on ice or frozen if possible). Samples must be sent to Dr. John Carlson, SEFSC Panama City Observer Program, 3500 Delwood Beach Road, Panama City, Florida 32408.
- Record the GPS location of where the animal was captured/released, the animal's condition at capture, a description of handling methods, and the condition at release.

For sea turtles:

- Observers must continue to tag and take tissue samples (under their ESA section 10 permit) from incidentally captured sea turtles. The SEFSC will be the clearinghouse for any genetic samples of sea turtles taken by observers.

For Atlantic and Gulf sturgeon:

- Refer to Appendix 2 for requirements for handling incidentally taken sturgeon and collecting genetic samples.

The following terms and conditions implement RPM 3:

F/SER2 and SEFSC must continue to review all data available on the observed/documented take of sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish in the shrimp fisheries and other suitable information (e.g., data on observed interactions with other fisheries, species distribution information, or surveys in the area where the fisheries operate) to assess whether there is sufficient information to undertake any additional analysis to attempt to identify correlations with environmental conditions or other drivers of incidental take within some or all of the action area. If such additional analysis is deemed warranted, it must be conducted. Within a reasonable amount of time after completing the review, F/SER2 and F/SER3 must consider taking appropriate action to reduce sea turtles, Atlantic and Gulf sturgeon, giant manta ray, and smalltooth sawfish interactions and/or their impacts.

The following terms and conditions implement RPM 4:

F/SER2 must remind fishers they are required to comply with our Sea Turtle Handling and Resuscitation Guidelines (Appendix 2) per regulations at 50 CFR 223.206(d)(1). F/SER2 must also disseminate the Giant Manta Ray Release Guidelines (Appendix 2) and the aforementioned Smalltooth Sawfish Handling, Release, and Reporting Procedures (Appendix 2) with federally-permitted vessels in the southeast U.S. shrimp fisheries (e.g., via Fishery Bulletin, NOAA website, etc.) within 6 months of issuance of this Opinion.

9 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs federal agencies to utilize their authority to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations identified in Opinions can assist action agencies in implementing their responsibilities under Section 7(a)(1). Conservation recommendations are discretionary activities designed to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The following conservation recommendations are discretionary measures that we believe are consistent with this obligation and, therefore, should be conducted or implemented by F/SER2:

1) F/SER2 should continue efforts with the states to establish consistent data protocols for fisheries effort and landings data between the Gulf of Mexico and South Atlantic regions.
2) FSER/2 should explore and support solutions and funding options to improve the electronic logbook program in the Gulf of Mexico shrimp fisheries and establish an electronic logbook program in the South Atlantic shrimp fisheries.
3) F/SER2 should request the SEFSC to investigate the efficacy of new TED designs for the otter trawl fisheries that would reduce the incidental bycatch and mortality of small sea turtles that would otherwise pass through the bars of currently required 4-in bar spacing.
4) F/SER2 should request the SEFSC to investigate the efficacy of TEDs in the skimmer trawl fisheries for vessels less than 40 ft in length.
5) F/SER2 should request the SEFSC to design a program for targeted electronic monitoring of the shrimp fisheries in areas where interactions of smalltooth sawfish are anticipated. The shrimp fisheries represent one of the most significant threats to this species, but specific data on the effects of the fisheries is lacking. Information on fisheries effort and species' presence may allow for discrete management efforts that could further reduce the effects of the fisheries on the smalltooth sawfish population and further recovery efforts.
6) F/SER2 should explore rulemaking to require the Sea Turtle Handling and Resuscitation Guidelines (Appendix 2) be posted inside the wheelhouse or an easily viewable area on the vessel if there is no wheelhouse for all federally-permitted shrimp trawlers, and all state-licensed shrimp trawlers, to the extent practicable. There are similar requirements for other fisheries (e.g., 50 CFR 622.29(a)(1)(i) in the Gulf reef fish fishery) to increase the survivability of captured and released sea turtles.
7) F/SER2 should create education and outreach material to communicate conservation messages for ESA-listed species, including materials for giant manta ray, through social media, websites, magazines, and print to federal agencies, local communities, and non-governmental organizations.

8) F/SER2 should recommend SEFSC staff explore additional in-water sea turtle research to document sea turtle movements, distribution, and habitat use that could help predict potential high-, medium-, and low-risk areas for fisheries interactions/bycatch.

10 REINITIATION OF CONSULTATION

This concludes formal consultation on the proposed actions. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where 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 incidental take is exceeded; (2) new information reveals effects of the agency action on listed species or designated critical habitat in a manner or to an extent not considered in this Opinion; (3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, F/SER2 must immediately request reinitiation of formal consultation and project activities may only resume if F/SER3 establishes that such continuation will not violate sections 7(a)(2) and 7(d) of the ESA. As first noted in Section 2.1, the lifespan of this Opinion is 10 years. Therefore, barring any other earlier need for reinitiation, a new Opinion on the proposed action will be necessary at the end of this 10-year period.

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APPENDIX 1 ANTICIPATED INCIDENTAL TAKE OF ESA-LISTED SPECIES IN FEDERAL FISHERIES

Table A.1. Anticipated Incidental Takes of Sea Turtles in Federal Fisheries (Greater Atlantic Region)

Fishery	ITS Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
American Lobster (July 31, 2014)	1 year	1 (lethal or nonlethal)	7 (lethal or nonlethal)	-	-	-
Batched Consultation[1] (gillnet; March 10, 2016)	5 years	1,345: no more than 835 lethal	4: no more than 3 lethal	4: no more than 3 lethal	4: no more than 3 lethal	-
Batched Consultation (bottom trawl; March 10, 2016)	4 years	852: no more than 284 lethal	4: no more than 2 lethal	3: no more than 2 lethal	3: no more than 2 lethal	-
Batched Consultation (trap/pot; March 10, 2016)	1 year	1 (lethal or nonlethal)	4 (lethal or nonlethal)	-	-	-
Atlantic Sea Scallop (dredge; November 27, 2018)	2 years	322: no more than 92 lethal	2 lethal (gears combined)	3: no more than 2 lethal (gears combined)	2 lethal (gears combined)	-
Atlantic Sea Scallop (trawl; November 27, 2018)	5 years	700; no more than 330 lethal				
Red Crab (February 6, 2002)	1 year	1 (lethal or nonlethal)	1 (lethal or nonlethal)	-	-	-

[1] Batched consultation includes the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass Fisheries

Table A.2. Anticipated Incidental Takes of Sea Turtles in Federal Fisheries (HMS)

Fishery	ITS Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
HMS, Excluding Pelagic Longline (January 10, 2020)	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 (May 15, 2020)	3 years	1,080: no more than 280 lethal	996: no more than 275 lethal	21: no more than 8 lethal in any combination		

Table A.3. Anticipated Incidental Takes of Sea Turtles in Federal Fisheries (Southeast Region)

Fishery	ITS Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
Caribbean Reef Fish (October 4, 2011)	3 years	None	18 (all lethal)	-	75 (all lethal)	51: no more than 3 lethal
Coastal Migratory Pelagics (November 18, 2017)	3 years	27 (7 lethal)	1 lethal	8 (2 lethal)	31 (9 lethal)	1 lethal
Dolphin-Wahoo (August 27, 2003)	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		
Gulf of Mexico Reef Fish (September 30, 2011)	3 years	1,044: no more than 572 lethal	11 lethal	108: no more than 41 lethal	116: no more than 75 lethal	9: no more than 8 lethal
Caribbean Spiny Lobster (December 12, 2011)	3 years	-	9 (lethal or nonlethal)	-	12 (lethal or nonlethal)	12 (lethal or nonlethal)
Gulf of Mexico/South Atlantic Spiny Lobster (August 27, 2009)	3 years	3 (lethal or nonlethal)	1 (lethal or nonlethal)		3 (lethal or nonlethal)	1 (lethal or nonlethal)
South Atlantic Snapper-Grouper (December 1, 2016)	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

Table A.4. Anticipated Incidental Take of Smalltooth Sawfish and Giant Manta Ray in Federal Fisheries

Fishery	ITS Period	Smalltooth Sawfish	Giant Manta Ray
HMS, Excluding Pelagic Longline (January 10, 2020)	3 years	23: no more than 1 lethal	9 nonlethal
Coastal Migratory Pelagics (November 18, 2017)	3 years	1 nonlethal	-
Gulf of Mexico/South Atlantic Spiny Lobster (August 27, 2009)	3 years	2 nonlethal	-
Gulf of Mexico Reef Fish (September 30, 2011)	3 years	8 nonlethal	-
South Atlantic Snapper-Grouper (December 1, 2016)	3 years	8 nonlethal	-

Table A.5. Anticipated Incidental Take of Atlantic Sturgeon by DPS in Federal Fisheries

Fishery	ITS Period	Atlantic Sturgeon DPS				
		Gulf of Maine	New York Bight	Chesapeake Bay	Carolina	South Atlantic
HMS, Excluding Pelagic Longline (January 10, 2020)	3 years	34: no more than 8 lethal	170: no more than 36 lethal	40: no more than 9 lethal	10: no more than 5 lethal	75: no more than 19 lethal
Coastal Migratory Pelagics (November 18, 2017)	3 years	2 nonlethal	4 nonlethal	3 nonlethal	4 nonlethal	10 nonlethal
Batched Consultation[1] (gillnet; March 10, 2016)	1 year (takes based on a 5-yr average)	137: no more than 17 lethal A.E.[2]	632: no more than 79 lethal A.E.	162: no more than 21 lethal A.E.	25: no more than 4 lethal A.E.	273: no more than 34 lethal A.E.
Batched Consultation[1] (bottom trawl; March 10, 2016)	1 year (takes based on a 5-yr average)	148: no more than 5 lethal A.E.	685: no more than 21 lethal A.E.	175: no more than 6 lethal A.E.	27: no more than 1 lethal A.E.	296: no more than 6 lethal A.E.
Atlantic Sea Scallop (dredge; November 27, 2018)	20 years	1 lethal (any DPS)				

[1] 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
[2] A.E.: adult equivalents

NOAA FISHERIES

Endangered Sawfish Handling, Release, and Reporting Procedures

for Individuals with Permitted Incidental Sawfish Take (commercial fishermen, non-targeted researchers, etc.)

An Endangered Species:

- Smalltooth sawfish are listed as endangered under the Endangered Species Act (ESA)
- Federal law prohibits injuring or harming sawfish
- Captured sawfish should be released immediately

Materials Needed:

- Measuring tape
- Net pick or boat hook
- Knife, line cutter, scissors
- Ropes
- Water quality meter (if available)
- Datasheets
- GPS
- Camera
- PIT reader (if available)

Reporting:

Adam Brame
Sawfish Recovery Coordinator
727-209-5958
Adam.Brame@noaa.gov

1-844-4SAWFISH

General Handling and Release Guidelines

- Work quickly to free and release the sawfish as soon as possible
- Keep sawfish in the water as much as possible, especially the gills
- Keep sawfish wet if it must be removed from the water
- Never remove the rostrum (saw)
- Do not stand or sit next to the rostrum
- Tie rope around tip of saw or tail only if needed to control sawfish for safety

Line Gear (longline, rod and reel, etc.) Specifics

- Keep the sawfish, especially the gills, in the water as much as possible
- Use line-cutting poles, long-handled dehookers, and/or boat hooks to remove line or gear
- Do not attempt to remove the hook, just cut the line as close to the hook as possible
- If line is tangled around the body or saw, untangle and remove as much of the line from around the sawfish as possible and then cut the line close to the hook

Net Gear (trawls, gillnets, etc.) Specifics

- Keep sawfish wet and in the net until ready for release
- Use line-cutting pole, scissors, and/or knife to cut free any net tangled around the saw by cutting the mesh along the length of the saw
- Once the mesh is cut, work it free with a boat hook or line-cutting pole

Data Recording

Please record as much information as you can quickly and safely including:

- Date and time
- Latitude and longitude (or detailed location description)
- Habitat description (water depth, temperature, salinity, dissolved oxygen)
- Photographs (in/on gear, body, rostrum)
- Markings, scars, wounds
- Tag number and type if applicable
- Lengths (saw and total, estimate if necessary)
- Sex
- Release condition including any remaining gear

U.S. Department of Commerce | National Oceanic and Atmospheric Administration | National Marine Fisheries Service

Requirements for Handling Incidentally Taken Sturgeon and Collecting Genetic Samples

General Handling of Sturgeon

1. If the animal appears energetic, active, and otherwise healthy enough to undergo handling, it should be done so in accordance with guideline #3 below. If the animal is not healthy enough to undergo the procedures described, ensure the vessel is in neutral and release it over the side, head first.
2. Animals should be handled rapidly, but with care and kept in water to the maximum extent possible during holding and handling. During handling procedures, the animal must be kept wet at all times using water from which it was removed (e.g., river water). While moving the animal or removing it from gear, covering its eyes with a wet towel may help calm it.
3. All handling procedures (i.e., measuring, PIT tagging, photographing, and tissue sampling) should be completed as quickly as possible, and should not exceed 20 minutes from when the sturgeon is first brought on board the vessel. Handling procedures should be prioritize in the following order: 1) collect a tissue sample (see procedure described below); 2) scan for existing PIT tags, apply new PIT tag if no pre-existing PIT tag is found; 3) measure the animal; and 4) photograph the animal. If all of the handling procedures cannot be completed within 20 minutes, the animal should be returned to the water; indicate which procedures were not completed when reporting the incidental take to NMFS.
4. A sturgeon maybe held on board for longer than 20 minutes only when held in a net pen/basket floating next to the vessel or placed in flow through tanks, where the total volume of water is replaced every 15-20 minutes.

Genetic Tissue Sampling for Atlantic Sturgeon

5. Genetic tissue samples must be taken from every Atlantic sturgeon captured unless conditions are such that collecting a sample would imperil human or animal safety.
6. Tissue samples should be a small (1.0 cm^2) fin clip collected from soft pelvic fin tissue. Use a knife, scalpel, or scissors that has been thoroughly cleaned and wiped with alcohol. Samples should be preserved in RNAlater™ preservative. Gently shake to ensure the solution covers the fin clip. Once the fin clip is in buffer solution, refrigeration/freezing is not required, but care should be taken not to expose the sample to excessive heat or intense sunlight. Label each sample with the fish's unique ID number. Do not use glass vials; a 2 ml screw top plastic vial is preferred (e.g., MidWest Scientific AVFS2002 and AVC100N).

PIT Tagging

7. Every sturgeon should be scanned for PIT tags along its entire body surface ensuring it has not been previously tagged. The PIT tag readers must be able to read both 125 kHz

and 134 kHz tags. When a previously implanted tag is detected the PIT tag information should be recorded on the reporting spreadsheet (Sturgeon Genetic Sample Submission Sheet). Indicate the animal was a recapture in the "comment" field of the reporting spreadsheet. A copy of that reporting spreadsheet should be sent to: mike_mangold@fws.gov.

8. Sturgeon without an existing PIT tag should have one implanted. The recommended frequency for PIT tags is 134.2 kHz. The tag information should be reported in the appropriate fields on the reporting spreadsheet.
9. Sturgeon smaller than 250 mm shall not be PIT tagged. Sturgeon measuring 250-350 mm TL shall only be tagged with 8 mm PIT tags. Sturgeon 350 mm or greater shall receive standard sized PIT tags (e.g., 11 or 14 mm).
10. PIT tags should be implanted to the left of the spine immediately anterior to the dorsal fin, and posterior to the dorsal scutes (Figure 1). This positioning optimizes the PIT tag's readability over the animal's lifetime. If necessary, to ensure tag retention and prevent harm or mortality to small juvenile sturgeon of all species, the PIT tag can also be inserted at the widest dorsal position just to the left of the 4th dorsal scute.
11. Scan the newly implanted tag following insertion to ensure it is readable before the animal is released. If the tag is not readable, one additional tag should be implanted on the opposite side following the same procedure, if doing so will not jeopardize the safety of the animal.

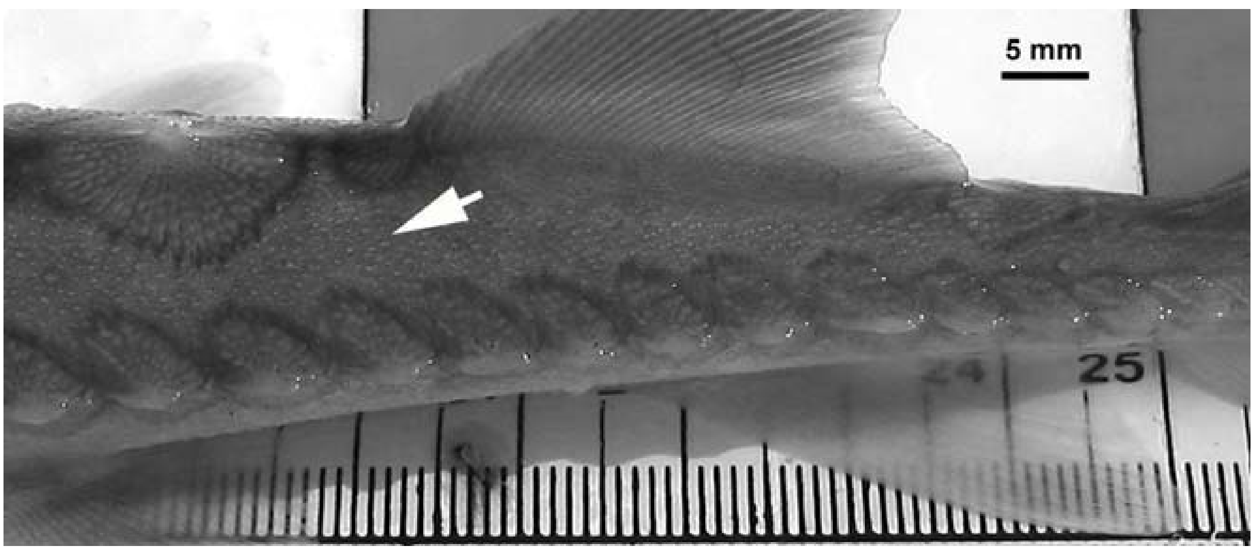


Figure 1. Standardized location for PIT tagging all Gulf, Atlantic, and shortnose sturgeon (Photo Credit: J. Henne, USFWS).

Measuring

12. Length measurements for all sturgeon should be taken as a straight-line measurement from the snout to the fork in the tail (i.e., fork length), and as a straight line measurement

from the snout to the tip of the tail (i.e., total length) (Figure 2). Do not measure the curve of the animal's body.

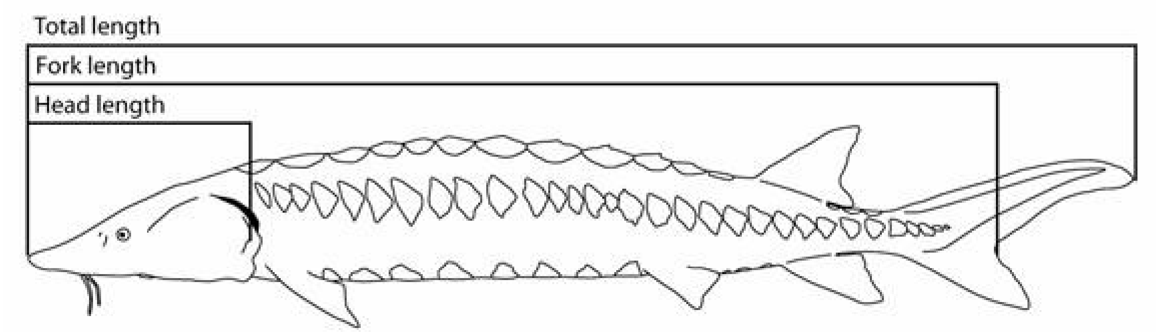


Figure 2. Diagram of different types of measurements for sturgeons (Drawing by Eric Hilton, Virginia Institute of Marine Science, in Mohead and Kahn 2010).

Reporting Captures/Samples

13. *Reporting Captures and Genetic Samples*: Incidental captures and genetic samples may be reported using the same reporting spreadsheet (Sturgeon Genetic Sample Submission Sheet). Electronic metadata for each sample must be provided to properly identify and archive samples. Submit the reporting spreadsheet via email to: rjohnson1@usgs.gov and takereport.nmfsser@noaa.gov. When submitting electronic metadata samples, identify the project name and biological opinion (SERO-2021-00087) in the subject line.
14. *Reporting Captures with NO Genetic Sample*: If no genetic sample could be safely collected, the incidental capture must still be reported using the Sturgeon Genetic Sample Submission Sheet. Submit the reporting spreadsheet via email to: takereport.nmfsser@noaa.gov. When submitting electronic metadata samples, identify the project name and biological opinion (SERO-2021-00087) in the subject line.

Transport of Genetic Samples

15. Package vials containing genetic samples together (e.g., in one box) with an absorbent material within a double-sealed container (e.g., zip lock baggie).
16. When submitting tissue samples via mail, identify the project name and biological opinion (SERO-2021-00087) under which the take was authorized in the shipping container. Ship tissue samples to:

    Robin Johnson
    U.S. Geological Survey
    Leetown Science Center
    Aquatic Ecology Branch
    11649 Leetown Road
    Kearneysville, WV 25430

Sea Turtle Handling and Resuscitation Requirements

NOAA FISHERIES

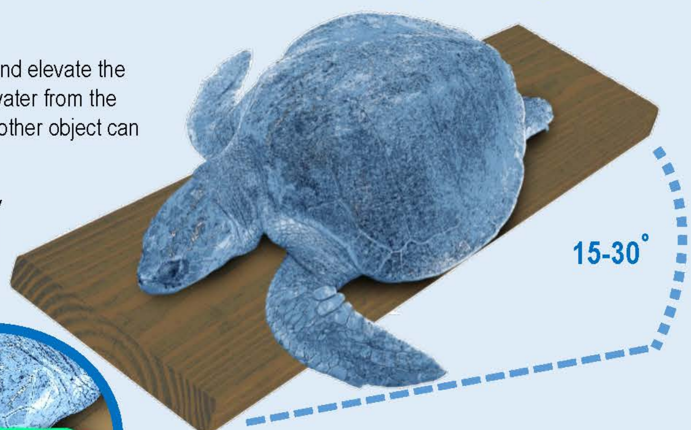
Per federal regulations at 50 CFR 223.206(d)(1):

Any sea turtle taken incidentally during fishing must be handled with care to prevent injury, evaluated to make sure it is active, and safely returned to the water.

Unresponsive turtles could still be alive and resuscitation must be attempted.

- Turtles that are unresponsive after capture may survive if allowed to recover.
- Sea turtles should only be considered dead if the muscles are stiff (rigor mortis), their body becomes bloated with gas, or the skin is detaching.

Resuscitation of unresponsive or inactive sea turtles must be attempted using the following procedures:



1. **Elevate Tail End:** Place the turtle right side up and elevate the hindquarters at least 6" (~15 - 30˚) to help drain water from the lungs. A board, tire, boat cushion, coiled rope, or other object can be used for elevation.
2. **Rock Gently:** Occasionally rock the turtle gently side to side by holding the outer edge of the shell and lifting one side about 3", then alternate to the other side.
3. **Check Eye Reflex:** Periodically, gently touch the corner of the eye or eyelid to see if the eyelid moves. This reflex will return as the turtle recovers.

4. **Keep Cool and Moist:** In warm weather (over 75˚F), keep the turtle shaded and moist. Place a water-soaked towel over the head, shell, and flippers or regularly wet the turtle with seawater to keep the turtle cool and moist. Never put the turtle into a container with water.
5. **Release Active Turtle Carefully:** Release active, resuscitated turtles as close to the water as possible. When doing so make sure fishing gear is not in use, the engine is in neutral, and avoid areas where the turtle may be recaptured or injured by other vessels.
6. **Give Them Time:** Attempt resuscitation for at least 4 hours. If there are no signs of life after 24 hours on deck, or if the muscles are stiff and/or the flesh has begun to rot, consider the turtle dead and return it to the water in the same manner (unless a NMFS observer retains the carcass).

Do not put the turtle on its back or pump the bottom shell (plastron) or try to force water out, as this is dangerous to the turtle.

Need assistance with a sea turtle or marine mammal in distress? *Call 844-SEA-TRTL (844-732-8785)*

Southeast Shrimp Fisheries Giant Manta Ray Release Guidelines

The guidelines presented here describe procedures for releasing a large ray from a shrimp trawl. Under these procedures, the trawl is retrieved in a normal manner and the ray is not brought onboard the vessel. The objective is to bring portions of the net tail and body out of the water in order to maneuver the captured ray towards and out the mouth of the net.

The capture of a manta ray during a tow often provides cues to the crew that should trigger net haulback. Once caught, large rays create an increase in the overall drag associated with the trawl. In some instances, the increase in drag, along with the rays thrashing against the trawl webbing, can provide noticeable cues. These cues can include an irregular "jerking" motion of the trawl cable above the water, a decrease in engine RPMs associated with an engine "lugging" sound, and a decrease in vessel speed. If the vessel is rigged for side trawling with outriggers, the vessel may veer off course and in the direction of the net that has captured the ray.


Photo: Josh Stewart

Step 1: The haulback of all nets should proceed as usual. Bring doors to the block.

Step 2: Position the vessel so that the manta/trawl is on the windward/upwind side of the boat. Reduce speed or take the engine out of gear if possible. This will reduce drag on the animal, allowing it to move towards the mouth of the net in subsequent steps.

Step 3: Retrieve the bag and dump the catch as usual.

Step 4: Using a whip/lifting line positioned forward of the TED, raise sections of trawl netting out of the water as high as possible, causing the animal to slide toward the trawl mouth.

- It may require several lifts/whips, moving forward in the trawl body with each lift, to move the animal toward the trawl mouth.
- If the animal stops moving at any point, try lowering the trawl doors to the water. This will increase the angle of the whip line lifting point relative to the trawl mouth and help move the animal toward the trawl mouth.

Step 5: If the animal does not move after repetitive lifts are attempted, it may be necessary to cut portions of the trawl net webbing that appear to be under tension near or around the animal. Bring those areas of the trawl as close to the vessel as possible and make necessary cuts to relieve tension. Take care to avoid cutting the animal.

Step 6: Once released from the trawl, monitor the animal's direction of movement. The ray may remain at the surface while it regains mobility. Take care to maneuver the vessel away from the animal while it is recovering.

Step 7: Report the incident to Calusa Horn, NMFS Southeast Giant Manta Ray Recovery Coordinator, at 727-824-5312, or *via* email Calusa.Horn@noaa.gov.

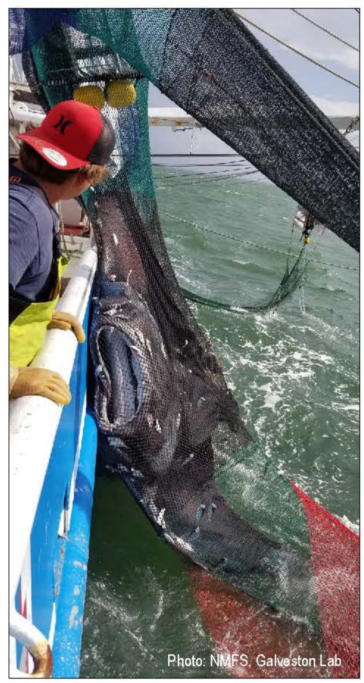


Photo: NMFS, Galveston Lab