UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Southeast Regional Office
263 13th Avenue South
St. Petersburg, Florida 33701-5505
https://www.fisheries.noaa.gov/region/southeast

12/09/2022

F/SER31:MCB
SERO-2022-02122

Jayson M. Hudson
Regulatory Project Manager
Policy Analyst Branch
Galveston District Corps of Engineers
Department of the Army
P.O. Box 1229
Galveston, Texas 77553-1229

Ref.: SWG-2019-00067, Port of Corpus Christi, Corpus Christi Ship Channel Deepening Project, Port Aransas, Nueces County, Texas

Dear Mr. Hudson:

The enclosed Biological Opinion (Opinion) on the referenced action responds to your request for consultation, pursuant to Section 7 of the ESA of 1973, as amended (16 U.S.C. § 1531 et seq.). We assigned the Opinion with a tracking number: SERO-2022-02122; please use this tracking number in all future correspondence related to this action.

This Opinion evaluates the effects of the proposed deepening of the Corpus Christi Ship Channel on threatened and endangered species and designated critical habitat, and is based on information provided by you and the published literature cited within. We conclude that the proposed action is likely to adversely affect but is not likely to jeopardize the continued existence of green (North Atlantic and South Atlantic Distinct Population Segments [DPS]), loggerhead (Northwest Atlantic DPS), and Kemp's ridley sea turtles, as well as giant manta ray.

We are providing an Incidental Take Statement (ITS) with this Opinion, which describes Reasonable and Prudent Measures that we consider necessary or appropriate to minimize the impact of incidental take associated with this action. The ITS also specifies Terms and Conditions, including monitoring and reporting requirements with which you and your applicants must comply.

We look forward to further cooperation with you on other projects to ensure the conservation of our threatened and endangered marine species and critical habitat. If you have any questions regarding this consultation, please contact Michael C. Barnette, Consultation Biologist, by phone at (727) 551-5794, or by email at michael.barnette@noaa.gov.

Sincerely,

AMENDOLA.KIMBERLY.BARBARA.1365830769
Digitally signed by AMENDOLA.KIMBERLY.BARBARA.1365830769
Date: 2022.12.09 13:15:39 -05'00'

for Andrew J. Strelcheck
Regional Administrator

Enclosure (s):
NMFS Biological Opinion SERO-2022-02122
cc: D. Klemm
D. Bernhart
nmfs.ser.esa.consultations@noaa.gov
File: 1514-22.f.8.

Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	U.S. Army Corps of Engineers, Galveston District
Activity:	Dredging and Disposal, Corpus Christi Ship Channel Deepening Project
Consulting Agency:	National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida NMFS Tracking Number SERO-2022-02122

AMENDOLA.KIMBERLY.BARBARA.1365830769 Digitally signed by AMENDOLA.KIMBERLY.BARBARA.1365830769 Date: 2022.12.09 13:16:08 -05'00'

Approved by:

for Andrew J. Strelcheck, Regional Administrator
NMFS, Southeast Regional Office
St. Petersburg, Florida

Date Issued: 12/09/2022

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Acronyms and Abbreviations

BIH	Brazos Island Harbor
BOEM	Bureau of Ocean Energy Management
CCL	curved carapace length
CCSC	Corpus Christi Ship Channel
CITES	Convention on International Trade in Endangered Species
CMP	coastal migratory pelagics
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
ECO	Environmental Consultation Organizer
EEZ	exclusive economic zone
ESA	Endangered Species Act
FAST-41	Title 41 of the Fixing America’s Surface Transportation Act
FMP	fishery management plan
FP	Fibropapillomatosis
FPISC	Federal Permitting Improvement Steering Council
FR	Federal Register
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
FY	Fiscal Year
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
LCS	large coastal sharks
LDWF	Louisiana Department of Wildlife and Fisheries
MLLW	mean lower low water
MMF	Marine Megafauna Foundation
MSA	mixed stock analysis
MSFCMA	Magnuson Stevens Fishery Conservation and Management Act
NA	North Atlantic (Ocean)
NAST	National Assessment Synthesis Team
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
ODESS	Operations and Dredging Endangered Species System
ODMDS	Ocean Dredged Material Disposal Site
OSHA	Occupational Safety and Health Administration
PAH	polycyclic aromatic hydrocarbons
PAIS	Padre Island National Seashore
PCB	polychlorinated biphenyls
PCCA	Port of Corpus Christi Authority
PFRU	Peninsular Florida Recovery Unit
PIM	post-interaction mortality
PLL	pelagic longline
PSO	protected species observer
PVA	population viability analysis
RPAs	reasonable and prudent alternatives
RPMs	reasonable and prudent measures
SA	South Atlantic (Ocean)
SCL	straight carapace length
SCS	small coastal shark
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
CY	cubic yards
ft	feet
hp	horsepower
in	inch
kg	kilogram
km	kilometer
kn	knots
L	liter
lb	pound
m	meter
mg	milligram
mi	miles
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 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., 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 proposed deepening of the CCSC on threatened and endangered species and designated critical habitat, in accordance with Section 7 of the ESA.

On July 5, 2022, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR Part 402 in 2019 ("2019 Regulations," see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order, and on November 14, 2022, the District Court issued an order remanding the regulations to the agencies without vacatur. As a result, the 2019 regulations are once again in effect, and we are applying the 2019 regulations here.

1 CONSULTATION HISTORY

The following is the consultation history for the NMFS ECO tracking number, SERO-2022-02122 CCSC Deepening Project.

On August 10, 2022, we received a biological assessment and request for formal consultation under Section 7 of the ESA from the USACE to permit dredging of the CCSC. The FPISC added the proposed CCSC project to the inventory of covered projects pursuant to the requirements set forth in FAST-41. We initiated formal consultation on August 11, 2022.

2 DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA

2.1 Proposed Action

The proposed action consists of deepening the CCSC (Figure 1) in 6 segments to -75 ft MLLW from the Gulf of Mexico to a station near Harbor Island, Texas, including the approximate 10-mi extension to the Entrance Channel necessary to reach sufficiently deep waters. Deepening would take place largely within the footprint of the currently authorized -54-ft MLLW channel. Dredging approximately 46.3 million CY over 5 years would be required with inshore and offshore placement of the material. Dredging would be conducted by both hopper and cutterhead dredges (Table 1). Dredged material would be placed in both inshore placement areas (with beneficial use objectives) and offshore at the ODMDS documented in Figure 2 below.

Table 1. CCSC Deepening Project Segments and Volume of Material to be Dredged.

Channel Segment	Year 1 Dredge Volume (CY)	Year 2 Dredge Volume (CY)	Year 3 Dredge Volume (CY)	Year 4 Dredge Volume (CY)	Year 5 Dredge Volume (CY)	Dredge Type
1	9,617,390	-	-	-	-	Hopper
2	-	10,154,381	10,154,381	-	-	Hopper
3	-	-	2,105,041	-	-	Hopper or Cutterhead
4	-	-	-	2,851,897	-	Cutterhead
5	-	-	-	2,951,614	-	Cutterhead
6	-	-	-	-	8,448,886	Cutterhead

USACE will employ measures to avoid and minimize adverse impacts to ESA-listed species during the proposed project. Specifically, these measures are:

1. Training: All contracted personnel involved in operating dredges may receive thorough training (as specified by NMFS or USFWS) on measures of dredge operation that will minimize impacts to listed species.
2. Observers: Typically, the PCCA would arrange for NMFS-approved PSOs to be aboard the hopper dredges to monitor the hopper bin, screening, and dragheads for sea turtles

and their remains. Observer coverage sufficient for 100% monitoring (i.e., 2 observers) of hopper dredging operations will be implemented.

3. Dredge Take Reporting: Observer reports of incidental take by hopper dredges would be submitted by email (takereport.nmfsser@noaa.gov) to SERO by onboard PSOs within 24 hours of any observed sea turtle take. Reports would contain information on location, start-up and completion dates, CY of material dredged, problems encountered, incidental takes, and sightings of protected species, mitigative actions taken, screening type, and daily water temperatures. An end-of-project summary report of the hopper dredging results and any documented sea turtle takes would be submitted to SERO within 30 working days of completion of the dredging project.
4. Seasonal Hopper Dredging Window: Hopper dredging activities would be completed between December 1 and March 31 if practicable, when sea turtle abundance is lower throughout Gulf coastal waters.
5. Sea Turtle Deflecting Draghead and Dredging Pumps: Typically, a state-of-the-art rigid deflector draghead would be used on hopper dredges at all times of the year. Typically, dredging pumps will be disengaged by the operator when the dragheads are not firmly on the bottom as indicated by sensors to prevent impingement or entrainment of sea turtles within the water column (especially important during dredging cleanup).
6. Non-hopper Type Dredging: Hydraulic or mechanical (bucket) dredges, which are not known to take turtles, may be used when possible between April 1 and November 30.
7. Cold Stunning Events: Vessel speed will be further reduced during cold weather events that are conducive to wildlife impacts. Occurrences of cold stunning events will be informed by PCCA participation in a regional group of experts led by academic professionals who model weather and water temperatures to give advance warning of potential cold stunning events. PCCA will also have a trained biologist on the vessel observing and monitoring for wildlife to stop operations accordingly during potential cold stunning events.
8. Dredge Lighting: From March 15 through October 1, sea turtle nesting and emergence season, all lighting aboard hopper dredges and support vessels operating within 3 nm of sea turtle nesting beaches would be limited to the minimal lighting necessary to comply with USCG and OSHA requirements. Non-essential lighting would be minimized through reduction, shielding, lowering, and appropriate placement.
9. Relocation Trawling: Typically, relocation trawling would be undertaken by a NMFS-approved protected species observer retained by the PCCA where any of the following conditions are met: (a) 2 or more turtles are taken in a 24-hour period in the project or, (b) 4 or more turtles are taken in the project. The purpose of the trawling would be to capture sea turtles that may be in the dredge path and relocate them away from the action area.
10. STSSN Notification: PCCA or its representative would notify the STSSN state representative of start-up and completion of dredging and relocation trawling operations. The STSSN would be notified of any turtle strandings in the project area that may bear the signs of interaction with a dredge. Stranded sea turtles would be reported to the Texas sea turtle hotline (1-866-TURTLE5 or 1-866-887-8535).

2.2 Action Area

The action area (Figure 1) for this consultation includes and is adjacent to Corpus Christi Bay, a 96,000-ac bay on the Texas central coast. The CCSC cuts through Corpus Christi Bay, which possesses an average depth of 11 ft, and extends past barrier islands and out into the Gulf of Mexico approximately 10 nm. The larger action area includes Nueces, San Patricio, Refugio, and Aransas Counties.

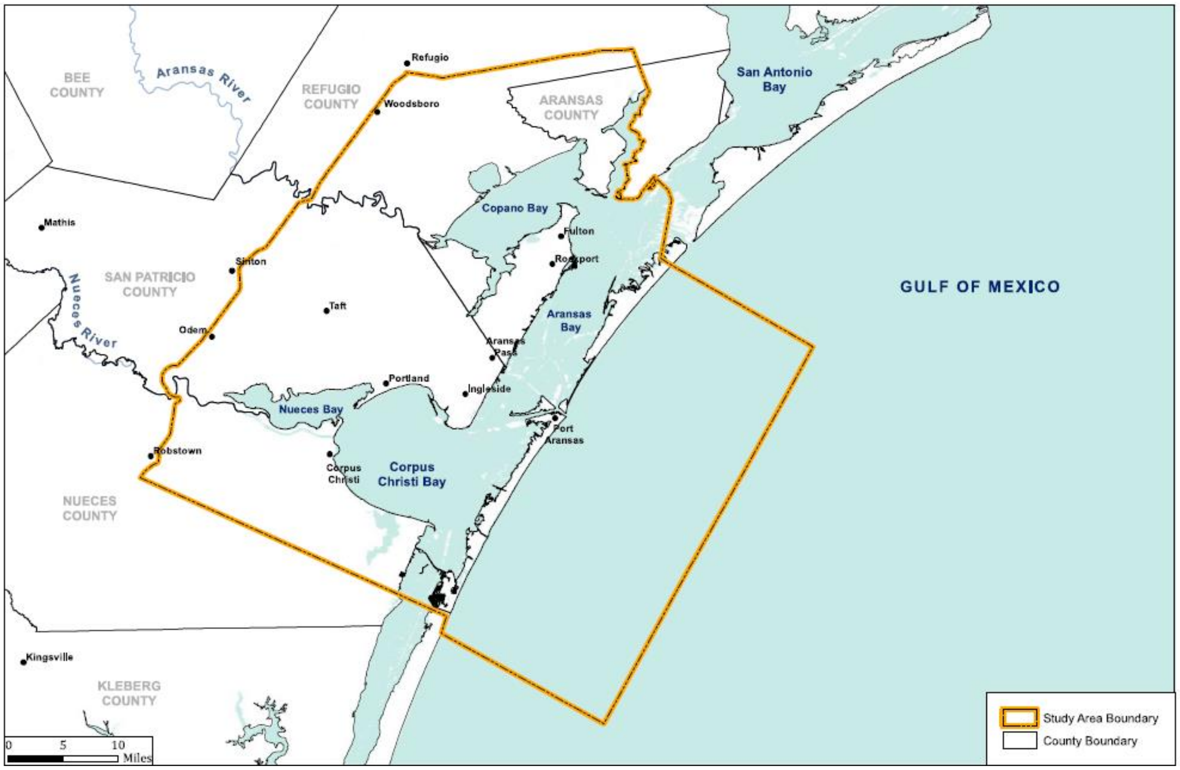

Figure 1. Map of the action area.

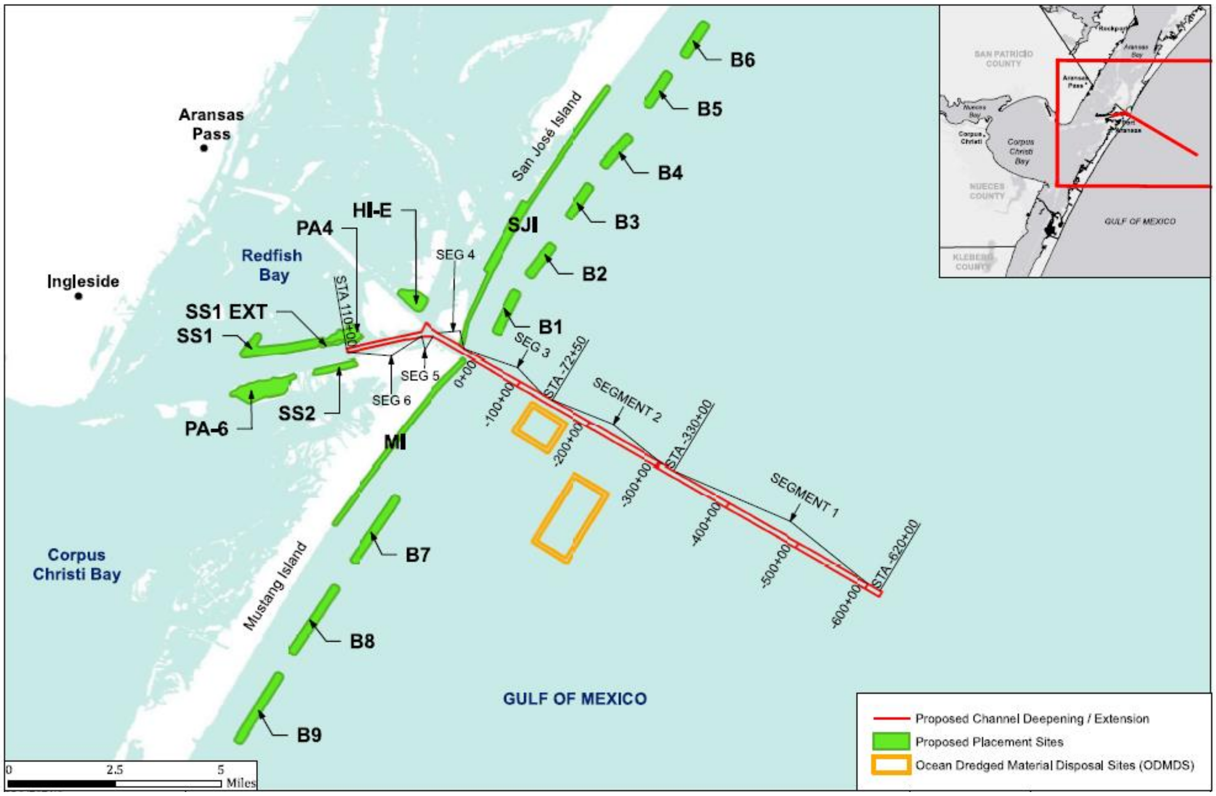

Figure 2. Map of the proposed project with placement areas (in green) and ODMDS sites (in orange).

2.3 Description of Primary Activities Conducted Under the Proposed Action

The primary activities conducted under the proposed action include dredging with hopper and cutterhead dredges, relocation trawling, and placement of dredged material. These activities are described in more detail below.

Hopper Dredging

A hopper dredge is a self-propelled ocean-going vessel with a section of the hull compartmented into 1 or more hoppers. Fitted with powerful pumps, the dredge sucks sediment from the surface of the seafloor through long intake pipes, called dragarms, and store it in the hoppers. Normal hopper dredge configuration has 2 dragarms, one on each side of the vessel. A dragarm is a pipe suspended over the side of the vessel with a suction opening called a draghead for contact with the bottom. Depending on the hopper dredge, a slurry of water and sediment is generated from the plowing of the draghead "teeth," the use of high pressure water jets, and the suction velocity of the pumps. The dredged slurry is distributed within the vessels hopper allowing for solids to settle out and the water portion of the slurry to be discharged from the vessel during operations through its overflow system. When the hopper attains a full load, dredging stops and the vessel travels to either an in-water placement site, where the dredged material is discharged through the bottom of the ship, or a site to hook up to an in-water pipeline, where the dredged material is transported to a shore placement site (e.g., beach nourishment).

Hopper dredges are well suited to dredging heavy sands. They can work in relatively rough seas but safety, effectiveness, and costs are a concern. Because they are mobile, they can be used in high-traffic areas. They are often used at ocean entrances and offshore, but cannot be used in confined or shallow areas due to their size and draft. Hopper dredges can move quickly to disposal sites under their own power (i.e., maximum speed unloaded ≤ 17 kn; maximum loaded ≤ 16 kn), but since the dredging stops during the transit to and from the disposal area, the operation loses efficiency if the haul distance is too far. Based on the review of hopper dredge speed data provided by the USACE Dredging Quality Management program, the average speed for hopper dredges while dredging is between 1-3 kn, with most dredges never exceeding 4 kn (NMFS 2020).

Hopper dredges also have several limitations. Considering their normal operating conditions, hopper dredges cannot dredge continuously unlike other dredge types that continue to work and transfer dredged material to another location. Hopper dredges must stop dredging while transporting materials to the final destination. The precision of hopper dredging is lower than other types of dredges; therefore, they have difficulty dredging steep side banks and cannot effectively dredge around structures. For example, dragheads may “crab” or move under or onto side slopes as a result of bottom conditions, bottom currents, or location of the dredge in or near the side of the channel. Crabbing may result in dragheads not being maintained on the bottom due to the more frequent need to pick up and realign the dragarms. Therefore, there is an increased risk of sea turtle entrainment when dredging within environments that may result in a higher risk of crabbing.

In order to minimize the risk of incidental takes of sea turtles, sea turtle deflectors are added to the dragheads used on hopper-dredging projects where the potential for sea turtle interactions exist and the dredging environment does not reduce the efficacy of the deflector or increase the risk for sea turtle interaction. The leading edge of the deflector is designed to have a plowing effect of at least 6-in depth when the drag head is being operated. Appropriate instrumentation is required on board the vessel to ensure that the critical “approach angle” is attained in order to satisfy the 6-in plowing depth requirement.

Cutterhead Dredging
Cutterhead dredges are designed to handle a wide range of materials including clay, hardpan, silts, sands, gravel, and some types of rock formations without blasting. They are used for new work and maintenance in projects where suitable placement/disposal areas are available and operate in an almost continuous dredging cycle resulting in maximum production, economy, and efficiency. Cutterhead dredges are capable of dredging in shallow or deep water and have accurate bottom and side slope cutting capability. A cutterhead is a mechanical device that has rotating blades or teeth to break up or loosen the bottom material so that it can be transported through a dredge pipeline. Cutterhead dredges require an extensive array of support equipment

including a pipeline (floating, shore, and submerged), boats (crew, work, survey), barges, and pipe handling equipment. Limitations of these dredges include relative lack of mobility, long mobilization and demobilization, inability to work in high wave action and currents, and they are impractical in high traffic areas.

During the dredging operation, a cutterhead dredge is held in position by 2 spuds at the stern of the dredge, only one of which can be on the bottom while the dredge swings. Some cutterhead dredges use a system of anchors and winches to hold themselves in place and/or advance forward. There are 2 swing anchors some distance from either side of the dredge, which are connected by wire rope to the swing winches. The dredge swings to port and starboard alternately, passing the cutter through the bottom material until the proper depth is achieved. The dredge advances by "walking" itself forward on the spuds. This is accomplished by swinging the dredge to the port, using the port spud and appropriate distance, then the starboard spud is dropped and the port spud raised. The dredge is then swung an equal distance to the starboard and the port spud is dropped and the starboard spud raised.

In most cases, dredged material is pumped directly from the dredged area to a placement/disposal site including using the aforementioned pipeline to transport the dredged material to an upland location or a barge for transport to a hydraulic off-load site. As such, there is no opportunity to monitor for biological material on board the dredge. Monitoring at the placement/disposal site is also challenging due to the volume of material pumped, often to the uplands, and often unsafe for an observer. Because the cutterhead is typically buried in the sediment to promote operational efficiency, exposure in the water column to the suction field is limited and cutterhead dredging has historically resulted in significantly lower takes of ESA-listed species than hopper dredging.

Relocation Trawling

Relocation trawling minimizes the risk of lethal encounters with a hopper dredging operation by intentionally capturing ESA-listed species to reduce the abundance those species in a project location. Modified shrimp trawling equipment is used to sweep the sea floor to either startle ESA-listed species out of the area, with open net relocation trawling, or to capture and often relocate these species, through the use of closed net relocation trawling. This management technique was originally initiated in the early 1980s at Canaveral Harbor, Florida (Rudloe 1981) and has continued to be used as a take minimization measure for dredging in the southeast.

Relocation trawling must maintain a safe distance from the hopper dredge and other vessel traffic in the area. Therefore, the trawler is often not working directly in front of the hopper dredge, but is instead continuously working to remove ESA-listed species from the general dredging area. Trawlers may sometimes need to leave the dredge footprint such as a navigation channel to avoid collision with vessels in the area. Relocation trawling vessels are also smaller than hopper dredges and therefore more restricted by the weather conditions in which they can safely operate. Relocation trawling will be used as part of the activities proposed, as described and limited by the avoidance/minimization measures proposed by the action agency. These would include the use of relocation trawling when: (a) 2 or more turtles are taken in a 24-hour period in the project, or (b) 4 or more turtles are taken in the project.

Placement of Dredged Material

As mentioned, dredged material would be placed in both inshore placement areas (with beneficial use objectives) and offshore at the ODMDS as documented in Figure 2. Beneficial use of dredged material is defined by USACE as "consistent with sound engineering practices and meets all federal environmental requirements, including those established under the Clean Water Act and the Marine Protection, Research, and Sanctuaries Act (see 33 CFR 335.7, 53 FR 14902)". The beneficial placement of material means that material dredged is able to be used for a desired purpose instead of a disposal site like an ODMDS.

The USACE considers beneficial use sites to include nearshore placement, placement alongside and downdrift of a navigation channel, and placement on a beach or other sandy habitat. Other beneficial uses include marsh creation, land creation, thin layer placement, fish and wildlife habitat enhancements, fisheries improvements, wetland restoration, etc.

The proposed action also includes the placement of material in the ODMDS sites identified by USACE and documented in Figure 2. The USACE informed us that expansion of the current ODMDS sites is not be required to complete the proposed dredging of CCSC described in the proposed action and, therefore, ODMDS expansion is not encompassed by this Opinion.

Vessel Traffic

The proposed action will employ a variety of vessels to complete the work including self-propelled hopper dredges, cutterhead dredges on barges tended by tug boats, crew and survey boats, and relocation trawlers. These vessels will largely be operating within the shipping channel and immediate adjacent areas, though hopper dredges will be transiting to and from ODMDS and other sediment placement areas noted in Figure 2.

3 EFFECTS DETERMINATIONS

Table 2 below documents listed species expected to occur within the action area, as well as action agency effects assessment; Table 3 documents listed critical habitat within the action area.

Table 2. Status of Listed Species that Potentially May Occur in the Action Area and Action Agency Effects Assessment (E=Endangered, T=Threatened, NLAA=Not Likely to be Adversely Affected, LAA=Likely to be Adversely Affected).

Species	Scientific Name	Status	Action Agency Effect Determination
Sperm whale	*Physeter macrocephalus*	E	NLAA
Giant manta ray	*Manta birostris*	T	NLAA
Loggerhead sea turtle, NWA DPS	*Caretta caretta*	T	LAA
Green sea turtle, NA and SA DPSs	*Chelonia mydas*	T	LAA
Leatherback sea turtle	*Dermochelys coriacea*	E	NLAA

Hawksbill sea turtle	*Eretmochelys imbricata*	E	LAA
Kemp's ridley sea turtle	*Lepidochelys kempii*	E	LAA

Table 3. Critical Habitat in the Action Area.

Species Critical Habitat	Critical Habitat Unit	Action Agency Effect Determination
Loggerhead sea turtle, NWA DPS	LOGG-S-2: Gulf of Mexico Sargassum	No Conclusion Provided

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

Sperm Whales

Sperm whales may be affected via vessel collisions with a dredge or other vessel associated with the proposed action. We believe this effect is extremely unlikely to occur. Sperm whales are predominantly found seaward of the continental shelf in waters distant from the proposed dredging of the CCSC. Sightings of sperm whales are almost exclusively in the continental shelf edge and continental slope areas (Scott and Sadove 1997). In the rare event that a listed whale is in the same vicinity of a dredge or other vessel associated with the proposed project, we expect the slow rate of vessel speed would give a whale or the vessel time to avoid a collision.

Leatherback Sea Turtle

We do not expect leatherback sea turtles to occur regularly within the action area. Leatherback sea turtles are pelagic typically and are found offshore. Sea turtle stranding records (i.e., traditional strandings excluding cold-stunning events) for the area (i.e., Neuces, San Patricio, and Aransas Counties) from 2012-2021 indicate leatherbacks are uncommonly encountered. During that time period, there were only 8 leatherback strandings documented out of over 3,900 stranding records (~0.2%). Of those 8 leatherback sea turtle stranding records, 4 were noted as severely decomposed, which could indicate they drifted for a significant period of time (i.e., from farther offshore). Only 1 leatherback sea turtle stranding was documented as alive. More directly relevant are the 161 sea turtle takes reported during USACE Galveston District dredging projects from 1995-2022, of which there have been no leatherback sea turtles documented. The lack of documented take is likely a result of the aforementioned pelagic habitat preference and the large size of leatherback sea turtles, which likely allow them to avoid entrainment by hopper dredges. As a result, we conclude the proposed action is not likely to adversely affect leatherback sea turtles.

Hawksbill Sea Turtle

While hawksbill sea turtles can be found in the action area, their presence is rare, as demonstrated by their less than 1% occurrence in sea turtle stranding reports (Table 7). Hawksbill sea turtles are closely associated with reef habitat and most prevalent in the Southeast Region in South Florida and the U.S. Caribbean. Because of their habitat presence, we do not believe hawksbill sea turtles will be entrained by the proposed hopper dredging. This is supported by the fact there are no reports of hawksbill sea turtles captured/taken during hopper dredging in the action area (or the larger Galveston District) from 1995-2022. Likewise, due to their general rarity within the action area (compared to other sea turtle species), we do not believe they will be captured by relocation trawling activities. In summary, we believe the proposed action is not likely to adversely affect hawksbill sea turtles.

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 proposed action occurs within *Sargassum* habitat, but we do not expect the proposed action will affect the primary constituent elements (i.e., 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.

3.2 Analysis of Potential Routes of Effects Not Likely to Adversely Affect Listed Species or Designated Critical Habitat

Effects Resulting from Cutterhead Dredge Interactions
Cutterhead dredges are a suction type dredge that operate when the cutterhead is generally embedded in sediment. The cutterhead creates a small zone of suction around the cutterhead; if the cutterhead were to be exposed to the water column and not completely embedded in sediment, it could entrain listed species.

Potential effects to sea turtles by cutterhead dredging include physical injury. We believe this route of effect is discountable based on information presented in other dredging Opinions (e.g., NMFS 2020). Specifically, we have only 1 documented sea turtle interaction with a cutterhead dredge, which was based on a live stranded green sea turtle discovered outside of the dredge discharge area with a cracked plastron and carapace. This stranding was 1 of 42 cold-stunned green sea turtle strandings during a cold front that swept through South Texas on December 22, 2004. Therefore, it cannot be linked definitively to injury caused by the cutterhead dredge. We have no other information or reported takes of sea turtles by cutterhead dredging, despite frequent use of cutterhead dredging within the action area and larger southeast U.S. Therefore, we believe the risk of physical injury or take of sea turtles by cutterhead dredging is an extremely unlikely event that we do not expect to occur during the proposed action. We continue to expect that sea turtles will move away from and avoid interaction with cutterhead dredging. Likewise, we believe the risk of injury to giant manta ray from cutterhead dredges is extremely unlikely due to their large size and ability to avoid the suction created by the cutterhead, pelagic lifestyle (i.e., versus benthic), and the location of planned cutterhead dredging operations close to shore or inshore (i.e., Segments 3-6 in Figure 2). In summary, we do not believe cutterhead dredge operations conducted under the proposed action will adversely affect sea turtles or giant manta ray.

Effects Resulting from Hopper Dredge Interactions
Hopper dredging may entrain or impinge giant manta ray; however, we believe these effects are extremely unlikely to occur. Giant manta ray are a large and extremely mobile species likely able to avoid the suction created by a hopper dredge. This conclusion is reinforced by our decades of experience with reporting of take from hopper dredging (i.e., since the 1980s), and a review of the available scientific literature, all of which document no known reports of hopper dredging entrainment of giant manta ray.

Hopper dredges are known to cause mortality to sea turtles, based on monitoring for sea turtle takes since 1980, by entrainment and impingement. We, therefore, believe that hopper dredging is likely to continue to adversely affect these species, as described below in Section 3.3, and as discussed in Section 6 of this Opinion.

Effects Resulting from Placement of Dredged Material
We believe that risk of a sea turtle or giant manta ray being caught in the discharge through the water column and buried on the sea floor is so low as to make the route of effect discountable. These mobile species would be able to detect the presence of the material being deposited and avoid being harmed by its placement. Placement in an open ocean environment such as an ODMDS or beneficial use site would allow room for species to move away from and around the placement. In addition, the presence of NMFS-approved PSOs (i.e., as required by this Opinion's Terms and Condition) allow for the monitoring of their presence, and activity will cease if they are detected in the immediate area.

Effects Resulting from Water Quality Issues
We believe changes in water quality resulting from turbidity from dredging and material placement analyzed under this Opinion may affect, but are not likely to adversely affect sea turtles or giant manta ray for the following reasons.

Cutterhead dredging may cause localized turbidity. Likewise, overflow from hopper dredging or from other equipment such as barges and scows could increase turbidity in the immediate area, and could likely cause a decrease in DO concentrations. We believe these effects on listed species will be insignificant. We expect that in open water environments these effects will be temporary, mobile species will avoid these disturbed areas, and turbidity will dissipate relatively quickly. Turbidity is not generally believed to impact sea turtles, as sea turtles breathe air and can therefore both move away from areas of poor water quality and surface to breathe air.

Effects from Vessel Traffic
The proposed action will include the use of several vessels including 2 hopper dredges, derrick and anchor barges for the cutterhead dredge, along with a tender and tow tug, crew and survey boats, and potentially relocation trawlers. ESA-listed species may be struck by these vessels during the proposed action. Sea turtles are air-breathing reptiles and may spend significant time at or near the water's surface, making them vulnerable to vessel strikes. We have STSSN data and other information documenting vessel impacts are a major source of mortality for sea turtles

in nearshore waters. Giant manta rays can be frequently observed traveling just below the surface and will often approach or show little fear toward humans or vessels (Coles 1916), which may also make them vulnerable to vessel strikes (Deakos 2010). However, information about interactions between vessels and giant manta rays is limited. We have at least some reports of vessel strike, including a report of 5 giant manta rays struck by vessels from 2016-2018; individuals had injuries (i.e., fresh or healed dorsal surface propeller scars) consistent with a vessel strike. These interactions were observed by researchers conducting surveys from Boynton Beach to Jupiter, Florida (J. Pate, Florida Manta Project, pers. comm. to M. Miller, NMFS OPR, 2018) and it is unknown where the manta was at the time of the vessel strike. This risk to sea turtles and giant manta ray is increased around inlets and shipping channels where vessel traffic will be more prevalent and some ESA-listed species may congregate (i.e., greater overlap of the risk and at-risk species).

We believe vessel traffic associated with the proposed action is extremely unlikely to affect ESA-listed species. While the proposed action will result in localized vessel traffic increases, given the significant ambient vessel traffic in the larger action area over the course of any given year, this increase will be insignificant. Further, only a small portion of anticipated vessel traffic will be conducted by hopper dredges to dispose of sediments (at ~15 kn). Vessel speeds for most of these vessels will be relatively slow (e.g., 5 kn). The hopper dredge has the potential to transit at approximately 15 kn to or from the ODMDS or other sediment placement areas, but typically it travels at 1-3 kn when actively dredging. In most cases, we believe sea turtles and giant manta ray have the ability and agility to move out of the way of vessels associated with the proposed action, should they be in the area. At this time, we are unaware of any sea turtles or giant manta ray identified with a vessel strike injury that have been directly related to dredging activities considered in any biological opinion (NMFS 2020).

3.3 Potential Routes of Effects Likely to Adversely Affect Listed Species

We anticipate that Kemp's ridley, green, and loggerhead sea turtles may be adversely affected by the proposed action due to the potential for hopper dredge take or capture by relocation trawler. We also anticipate that giant manta ray may be adversely affected by capture in the relocation trawler. A detailed discussion on these effects is included in Section 6.

Effects Resulting from Hopper Dredge Interactions
Hopper dredges are known to cause mortality to sea turtles, based on monitoring for sea turtle takes since 1980, by entrainment and impingement. We, therefore, believe that hopper dredging is likely to continue to adversely affect these species, as described below and discussed in Section 6 of this Opinion. Species can become entrained in hopper dredges as the draghead moves along the bottom. Entrainment occurs when the species cannot escape from the suction of the dredge and they are sucked into the dredge draghead, pumped through the intake pipe, and then killed as they cycle through the centrifugal pump and into the hopper. Because entrainment is believed to occur primarily while the draghead is operating on the bottom, it is likely that only those species feeding or resting on or near the bottom would be vulnerable to entrainment. They

can also be entrained if suction is created in the draghead by current flow while the device is being placed or removed, or if the dredge is operating on an uneven or rocky substrate and rises off the bottom. Recent information from the USACE suggests that the risk of entrainment is highest when the bottom terrain is uneven or when the dredge is conducting "cleanup" operations at the end of a dredge cycle when the bottom is trenched and the dredge is working to level out the bottom. In these instances, it is difficult for the dredge operator to keep the draghead buried in the sediment, thus species near the bottom may be more vulnerable to entrainment. Sea turtles resting in deeper waters or holes in the channel may be at an increased risk of take from dredging activities conducted there. Species can also be crushed on the bottom by the moving draghead and not entrained.

Effects Resulting from Relocation Trawling
Relocation trawling is used to minimize the risk of lethal hopper dredging take by sweeping the area around a hopper dredge with modified shrimp trawl nets to capture and relocate ESA-listed species that may be in the dredging area. While relocation trawling is intended to reduce the occurrence of lethal take from hopper dredging, the process of relocating ESA-listed species is, in and of itself, a form of take under the ESA for those species that are caught. Relocation trawling covered under this Opinion will be monitored by observers based on the guidance provided in the Terms and Conditions in Section 9.4, and includes handling and reporting guidance for ESA-listed species captured during relocation trawling. Additional relocation trawling parameters limit tow times to 42 minutes (though 30 minute tows are typical) to minimize the risk of adverse effects on ESA-listed species, primarily mortality of sea turtles due to forced submergence (NRC 1990; Epperly et al. 2002).

A study of the effects of relocation trawling as a mitigation tool to minimize the risk of take from hopper dredging (Dickerson et al. 2008) and data provided by the USACE on relocation trawling take in their ODESS demonstrate both the risk and benefits of this method. The risks to ESA-listed species of directed take are the stress endured by these species in the process of being trawled and relocated, including any potential physical harm during this process and stress that may result in reduced fitness in the form of reduced foraging and reproductive success. Relocation trawling may also have varying levels of effectiveness as a minimization of take with hopper dredging depending on the timing, trawling effort, and project location features. In Section 6 of this Opinion, we consider these effects to species relocated in the action area.

4 STATUS OF ESA-LISTED SPECIES CONSIDERED FOR FURTHER ANALYSIS

4.1 Sea Turtles

4.1.1 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 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, 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 2020). 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 nourishment, 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., DDT, 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 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., juvenile loggerhead and green sea 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.

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.

4.1.2 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 3), 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 data, 2019). Nesting numbers rebounded in 2020 (18,068 nests) and 2021 (17,671 nests) (CONAMP data, 2021). 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, rebounding to 262 nests in 2020, and back to 195 nests in 2021 (NPS data).

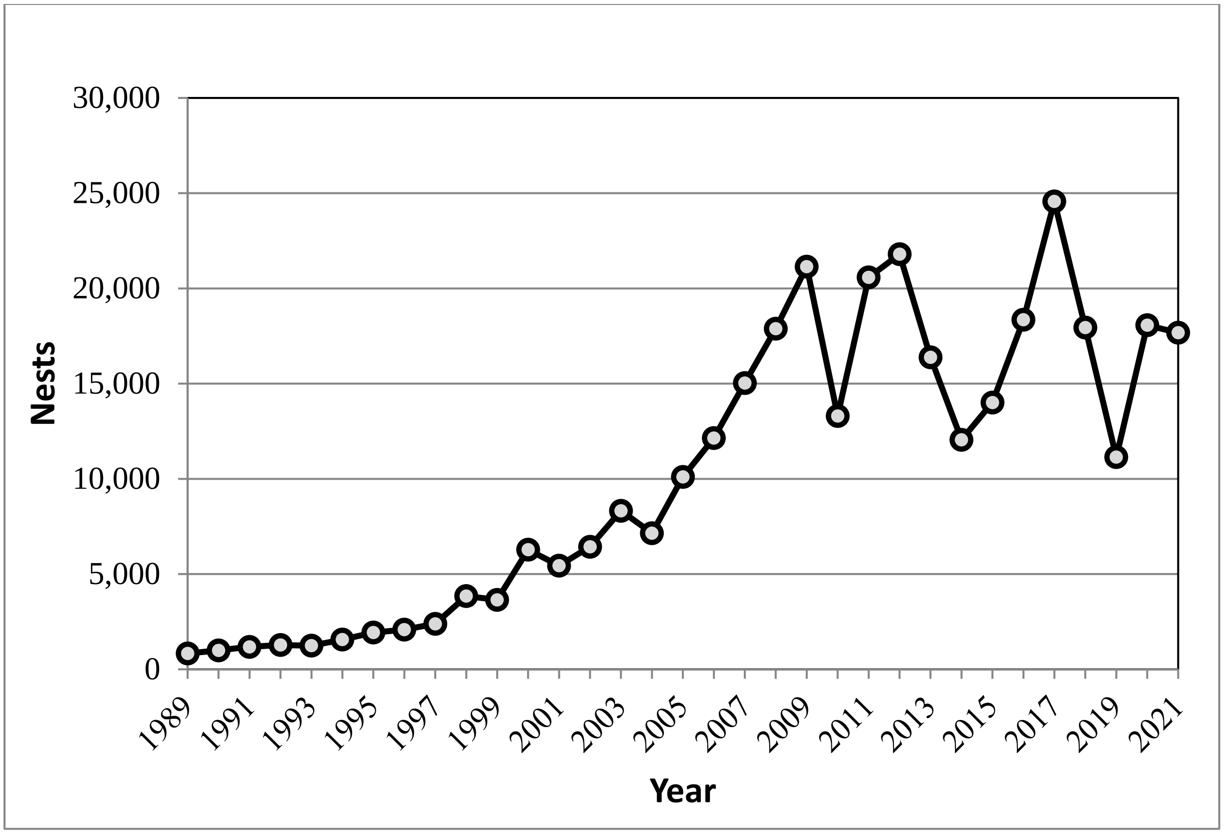

Figure 3. Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019).

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 4.1.1; 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* are increasingly established, bacterial and fungal pathogens in nests are also likely to increase; *arribada* is the Spanish word for "arrival" and is the term used for massive synchronized nesting within the genus *Lepidochelys.* 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 STSSN data) elevated sea turtle strandings in the Northern Gulf of Mexico, particularly throughout the Mississippi Sound area. For example, in the first 3 weeks of June 2010, over 120 sea turtle strandings were reported from Mississippi and Alabama waters, none of which exhibited any signs of external oiling to indicate effects associated with the DWH oil spill event. A total of 644 sea turtle strandings were reported in 2010 from Louisiana, Mississippi, and Alabama waters, 561 (87%) of which were Kemp's ridley sea turtles. During March through May of 2011, 267 sea turtle strandings were reported from Mississippi and Alabama waters alone. A total of 525 sea turtle strandings were reported in 2011 from Louisiana, Mississippi, and Alabama waters, with the majority (455) having occurred from March through July, 390 (86%) of which were Kemp's ridley sea turtles. During 2012, a total of 384 sea turtles were reported from Louisiana, Mississippi, and Alabama waters. Of these reported strandings, 343 (89%) were Kemp's ridley sea turtles. During 2014, a total of 285 sea turtles were reported from Louisiana, Mississippi, and Alabama waters, though the data is incomplete. Of these reported strandings, 229 (80%) were Kemp's ridley sea turtles. These stranding numbers are significantly greater than reported in past years; Louisiana, Mississippi, and Alabama waters reported 42 and 73 sea turtle strandings for 2008 and 2009, respectively. It should be noted that stranding coverage has increased considerably due to the DWH oil spill event.

Nonetheless, considering that strandings typically represent only a small fraction of actual mortality, these stranding events potentially represent a serious impact to the recovery and survival of the local sea turtle populations. While a definitive cause for these strandings has not been identified, necropsy results indicate a significant number of stranded turtles from these events likely perished due to forced submergence, which is commonly associated with fishery interactions (B. Stacy, NMFS, pers. comm. to M. Barnette, NMFS PRD, March 2012). Yet, available information indicates fishery effort was extremely limited during the stranding events. The fact that 80% or more of all Louisiana, Mississippi, and Alabama stranded sea turtles in the past 5 years were Kemp's ridleys is notable; however, this could simply be a function of the 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 4.1.1, 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.

4.1.3 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 4). 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 2 DPSs with individuals occurring in the Atlantic and Gulf of Mexico waters of the United States.

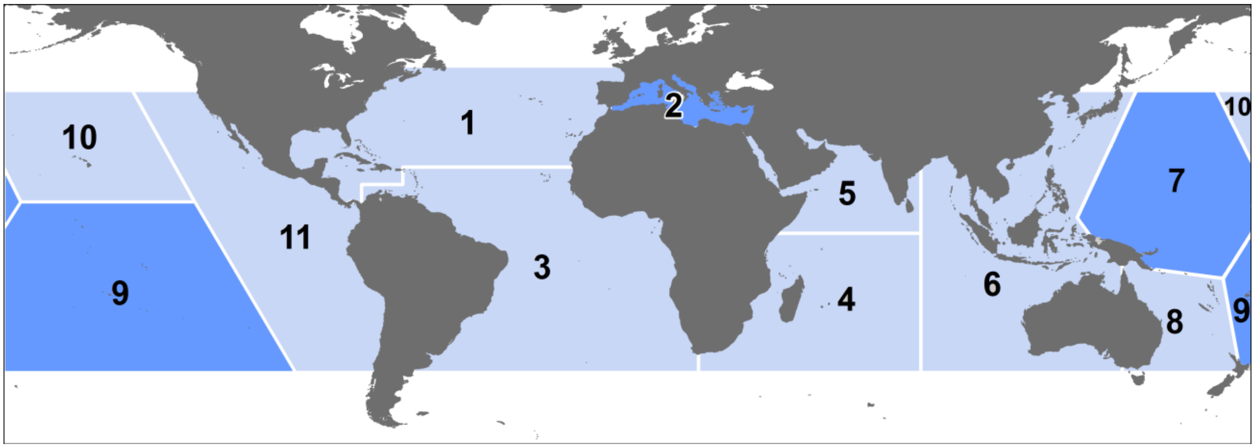

Figure 4. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic (NA); 2. Mediterranean; 3. South Atlantic (SA); 4. Southwest Indian; 5. North Indian; 6. East Indian-West Pacific; 7. Central West Pacific; 8. Southwest Pacific; 9. Central South Pacific; 10. Central North Pacific; 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, 2 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 4. 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 4, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into 4 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). 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. 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 5). According to data collected from Florida's index nesting beach survey from 1989-2021, 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. The pattern departed from the low lows and high peaks in 2020 and 2021 as well, when 2020 nesting only dropped by half from the 2019 high, while 2021 nesting only increased by a small amount over the 2020 nesting (Figure 5).

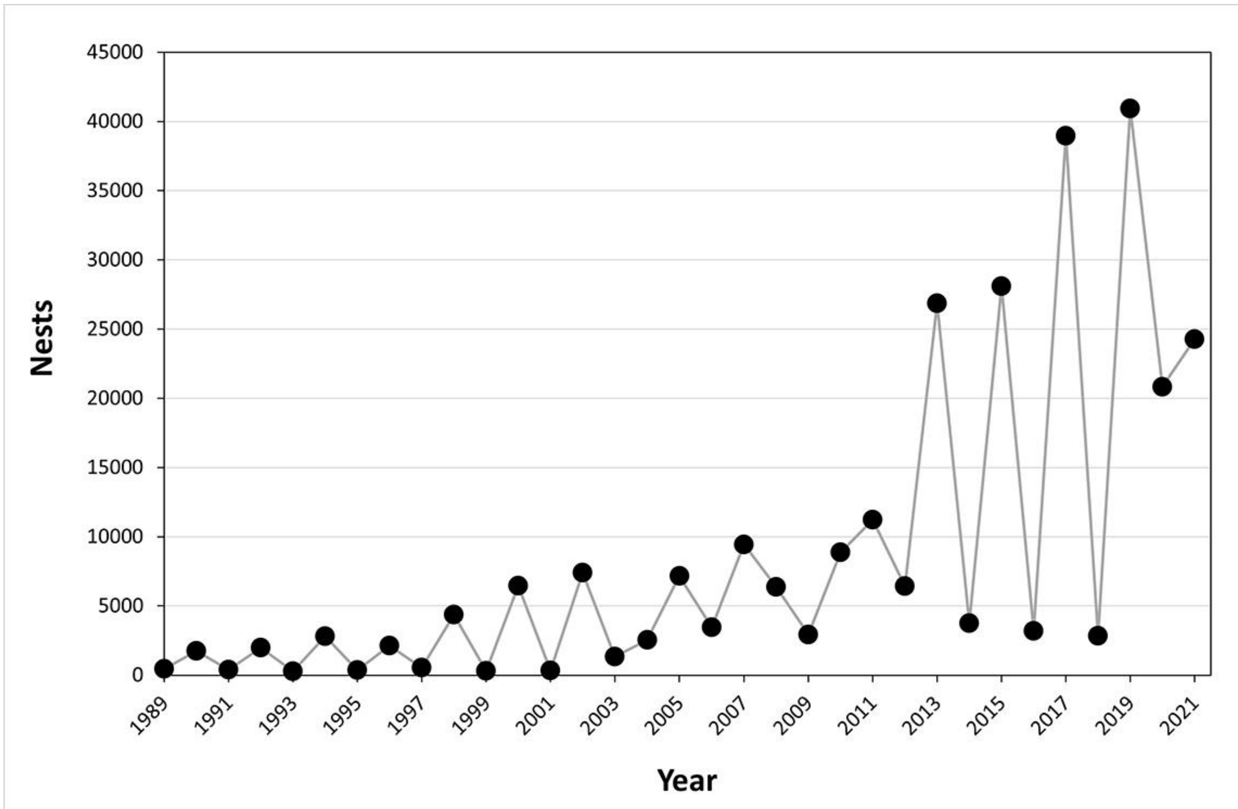

Figure 5. Green sea turtle nesting at Florida index beaches since 1989.

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 4.1.1.

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 4.1.1, 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).

4.1.4 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 NWA 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 [neritic refers to the nearshore marine environment from the surface to the sea floor where water depths do not exceed 200 m]); 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 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 2009; NMFS 2001; NMFS and USFWS 2008; TEWG 1998; TEWG 2000; TEWG 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

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

Peninsular Florida Recovery Unit

The 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 2020 was 105,164 nests (FWRI nesting database).

In addition to the total nest count estimates, 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 6). 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. While nest numbers subsequently declined from the 2016 high FWRI noted that the 2007-2021 period represents a period of increase. 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 again each year through 2020, reaching 53,443 nests before dipping back to 49,100 in 2021. 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).

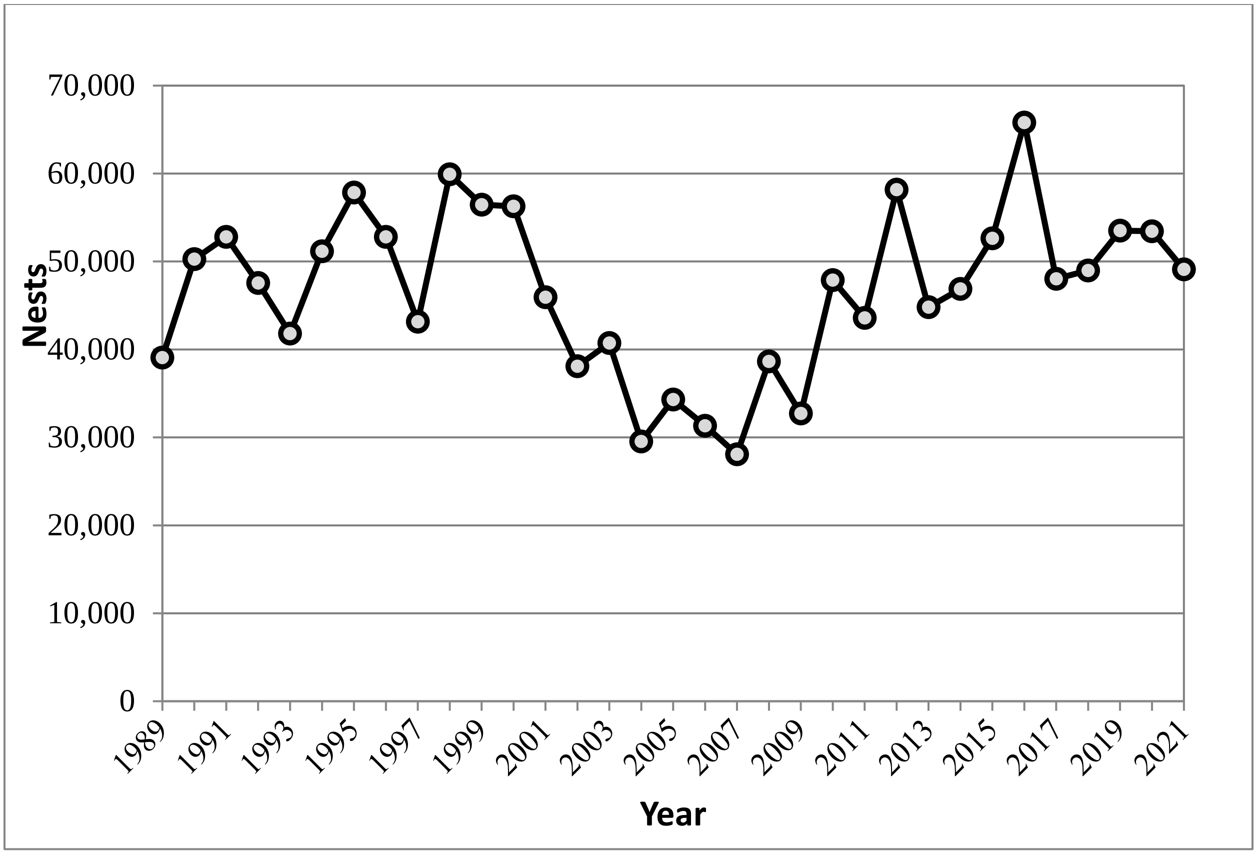

Figure 6. Loggerhead sea turtle nesting at Florida index beaches since 1989.

<u>Northern Recovery Unit</u>

Annual nest totals from beaches within the NRU averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches (GADNR unpublished data, 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 4) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, GADNR press release, http://www.georgiawildlife.com/node/3139). South Carolina and North Carolina nesting have also begun to shift away from the past declining trend. Loggerhead nesting in Georgia, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016. Nesting in 2017 and 2018 declined relative to 2016, back to levels seen in 2013 to 2015, but then bounced back in 2019, breaking records for each of the 3 states and the overall recovery unit. Nesting in 2020 and 2021 declined from the 2019 records, but still remained high, representing the third and fourth highest total numbers for the NRU since 2008.

Table 4. Total Number of NRU Loggerhead Nests (GADNR, SCDNR, and NCWRC nesting datasets compiled at Seaturtle.org).

Nests Recorded				
Year	**Georgia**	**South Carolina**	**North Carolina**	**Totals**
2008	1,649	4,500	841	**6,990**
2009	998	2,182	302	**3,472**
2010	1,760	3,141	856	**5,757**
2011	1,992	4,015	950	**6,957**
2012	2,241	4,615	1,074	**7,930**
2013	2,289	5,193	1,260	**8,742**
2014	1,196	2,083	542	**3,821**
2015	2,319	5,104	1,254	**8,677**
2016	3,265	6,443	1,612	**11,320**
2017	2,155	5,232	1,195	**8,582**
2018	1,735	2,762	765	**5,262**
2019	3,945	8,774	2,291	**15,010**
2020	2,786	5,551	1,335	**9,672**
2021	2,493	5,639	1,448	**9,580**

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. After another drop in 2018, a new record was set for the 2019 season, with a return to 2016 levels in 2020 and 2021 (Figure 7).

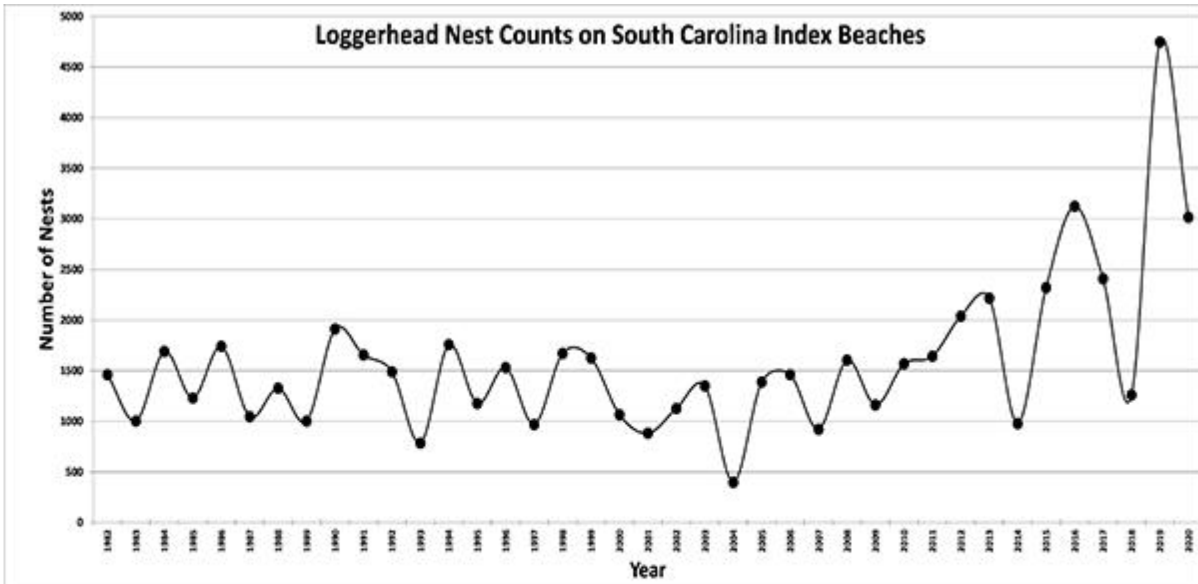

Figure 7. South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: http://www.dnr.sc.gov/seaturtle/ibs.htm).

Other NWA DPS Recovery Units

The remaining 3 recovery units—(DTRU, NGMRU, and 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. From 1989-2018 the average number of NGMRU nests annually on index beaches was 169 nests, with an average of 1,100 counted in the statewide nesting counts (Ceriani et al. 2019). Nesting survey effort has been inconsistent among the GCRU nesting beaches, and no trend can be determined for this subpopulation (NMFS and USFWS 2008). Zurita et al. (2003) found a statistically significant increase in the number of nests on 7 of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. Nonetheless, nesting has declined since 2001, and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008).

In-water Trends

Nesting data are the best current indicator of sea turtle population trends, but in-water data also provide some insight. In-water research suggests the abundance of neritic juvenile loggerheads is steady or increasing. Although Ehrhart et al. (2007) found no significant regression-line trend in a long-term dataset, researchers have observed notable increases in CPUE (Arendt et al. 2009; Ehrhart et al. 2007; Epperly et al. 2007). Researchers believe that this increase in CPUE is likely linked to an increase in juvenile abundance, although it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. Bjorndal et al. (2005), cited in NMFS and USFWS (2008), caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern 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 2009). The model uses the range of published information for the various parameters including mortality by stage, stage duration (years in a stage), and fecundity parameters such as eggs per nest, nests per nesting female, hatchling emergence success, sex ratio, and remigration interval. Resulting trajectories of model runs for each individual recovery unit, and the western North Atlantic population as a whole, were found to be very similar. The model run estimates from the adult female population size for the western North Atlantic (from the 2004-2008 time frame), suggest the adult female population size is approximately 20,000-40,000 individuals, with a low likelihood of females' numbering up to 70,000 (NMFS 2009). A less robust estimate for total benthic females in the western North Atlantic was also obtained, yielding approximately 30,000-300,000 individuals, up to less than 1 million (NMFS 2009). A preliminary regional abundance survey of loggerheads within the northwestern Atlantic continental shelf for positively identified loggerhead in all strata estimated about 588,000 loggerheads (interquartile range of 382,000-817,000). When correcting for unidentified turtles in proportion to the ratio of identified turtles, the estimate increased to about 801,000 loggerheads (interquartile range of 521,000-1,111,000) (NMFS 2011).

Threats
The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 4.1.1. 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 4.1.1, 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.

4.2 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 8) (Kashiwagi et al. 2011; Marshall et al. 2009), as may occasionally occur within estuaries (e.g., lagoons and bays).

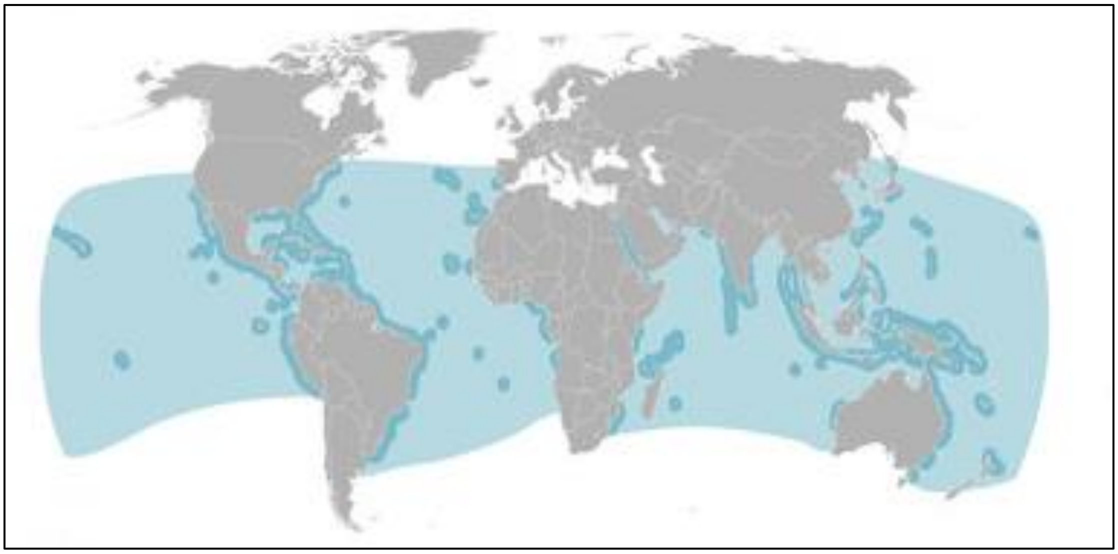

Figure 8. The Extent of Occurrence (dark blue) and Area of Occupancy (light blue) based on species distribution (Lawson et al. 2017).

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. 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 MMF 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).

5 ENVIRONMENTAL BASELINE

This section describes the effects of past and ongoing human and natural factors contributing to the current status of the species, their habitats, and ecosystem within the action area without the additional effects of the proposed action. In the case of ongoing actions, this section includes the effects that may contribute to the projected future status of the species, their habitats, and ecosystem. The environmental baseline describes the species' health based on information available at the time of the consultation.

By regulation, the environmental baseline for an Opinion refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early Section 7 consultation, and the impact of State or private actions, which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR 402.02).

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

5.1 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 4. As stated in Section 2.2, the proposed action would occur along the Texas central coast adjacent to Corpus Christi Bay.

5.2 Factors Affecting Listed Species within the Action Area

5.2.1 Federal Actions

We have undertaken a number of Section 7 consultations to address the effects of federally managed 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.

5.2.1.1 Fisheries

Within the action area, both recreational and commercial fisheries occur in state and federal waters. Globally, 6.4 million tons of fishing gear is lost in the oceans every year (Wilcox et al. 2015). Lost traps and disposed monofilament and other fishing lines are a documented source of mortality in sea turtles due to entanglement that may anchor an animal to the bottom. Materials entangled tightly around a body part may cut into tissues, enable infection, and severely compromise an individual’s health (Derraik 2002). Entanglements also make animals more vulnerable to additional threats (e.g., predation and vessel strikes) by restricting agility and swimming speed. The majority of ESA-species that die from entanglement in fishing gear likely sink at sea rather than strand ashore, making it difficult to accurately determine the extent of such mortalities.

Fishery interaction remains a major factor in sea turtle recovery and, frequently, the lack thereof. Wallace et al. (2010) estimated that worldwide, 447,000 sea turtles are killed each year from bycatch in commercial fisheries. In the most recent Opinion on the Southeastern U.S. shrimp fisheries, we estimate 17,010 Kemp’s ridley, 4,300 loggerhead, 3,400 green, 10 leatherback, and 10 hawksbill sea turtle mortalities over the next 10 years (NMFS 2021); this includes mortalities resulting from bycatch occurring in both state and federal waters. Although TEDs and other bycatch reduction devices have significantly reduced the level of bycatch to sea turtles and other marine species in U.S. waters, mortality still occurs. Giant manta ray are also caught as bycatch in fisheries.

In addition to commercial bycatch, recreational hook-and-line interactions also occur. Stacy et al. (2020) analyzed Texas sea turtle stranding data and determined evidence of fishing tackle/gear hooking injuries and entanglement in stranded turtles varied by species and stranding zone. For instance, evidence of fishing tackle/gear on stranded turtles were documented in 42.6% of stranded green sea turtles in Zone 20 (which includes Aransas Pass), but only 29.8% in Zone 21 to the south (Stacy et al. 2020). The authors concluded that presence of fishing tackle/gear injuries in stranded turtles was directly correlated to the proximity of inlets. Recreational hook-and-line interactions have also been documented with giant manta ray.

Fisheries in federal waters have been the subject of multiple Section 7 consultations in the action area and beyond. These fisheries include gillnet, longline, other types of hook-and-line gear, trawl gear, and pot fisheries. As described in Section 4 of this Opinion, available information suggests that mobile ESA-listed species can be captured in these gear types when the operation of the gear overlaps with the distribution of the species. For all fisheries for which there is a federal FMP, or for which any federal action has been taken to manage that fishery, impacts have been evaluated under Section 7. Formal Section 7 consultations have been conducted on the following fisheries, occurring at least in part within the action area, found likely to adversely affect threatened and endangered species: Atlantic shark fisheries, coastal migratory pelagic, and Southeast shrimp trawl fisheries. An ITS has been issued for the take in each of these fisheries. None of the Opinions for these fisheries concluded that the fisheries at issue were likely to jeopardize ESA-listed species or adversely modify designated critical habitat. Detailed information regarding the effects of each fishery can be found in the respective Opinions.

5.2.1.2 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 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.

To reduce take of listed species, relocation trawling may be utilized to capture and move sea turtles. In relocation trawling, a boat equipped with nets precedes the dredge to capture sea turtles 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.

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. In summary, dredging and disposal to maintain navigation channels, and removal of sediments for beach nourishment occurs frequently and throughout the range of sea turtles annually. This activity has, and continues to, threaten the species.

We originally completed regional Opinions on the impacts of USACE’s hopper-dredging operation in 2003 for operations in the Gulf of Mexico (i.e., GRBO). We revised the GRBO in 2007 (NMFS 2007a), 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 Opinion considers maintenance dredging and sand mining operations. We have produced numerous other "free-standing" Opinions that analyzed the impacts of 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 this regional Opinion. Any free-standing Opinion 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.

5.2.1.3 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 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.

5.2.1.4 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 2007b) have concluded that sea turtle takes may also result from marine debris and oil spills.

5.2.1.6 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 because of the potential impacts of recreational fishing from these piers on these species. For instance, in 2017 we consulted on the Quintana County Pier in Brazoria County, Texas, which concluded the action would not jeopardize listed species and provided an ITS for 3-year takes of 24 sea turtles (20 Kemp's ridley, 2 green, and 2 loggerhead sea turtles). We have conducted other similar pier consultations in Texas and throughout the larger Gulf of Mexico region.

5.2.1.7 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) non-lethal.

5.2.2 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 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. Additional discussion on some of these activities follows.

5.2.2.1 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.

Trawl Fisheries
Trawl fisheries, such as ones operating for shrimp, blue crab, and sheepshead, may also interact with sea turtles 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 non-shrimp trawl fisheries.

Recreational Fishing
Recreational fishing from private vessels may occur in the action area, and these activities may interact with sea turtles 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 5.2.1.1), 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 TEWG reports (TEWG 1998; TEWG 2000).

5.2.2.2 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 (https://www.fisheries.noaa.gov/national/marine-life-distress/sea-turtle-stranding-and-salvage-network). 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.

5.2.2.3 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.

5.2.3 Other Potential Sources of Impacts to the Environmental Baseline

5.2.3.1 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.

5.2.3.2 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/L) 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 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.

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.

5.2.4 Conservation and Recovery Actions Shaping the Environmental Baseline

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.

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

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 USCG, or any other federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. We already afford the same protection to sea turtles listed as threatened under the ESA (50 CFR 223.206(b)).

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.

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

5.3 Summary

In summary, several factors adversely affect sea turtles and giant manta ray 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, 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 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. 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 have significantly increased over the past 10 years and are expected to continue.

5.4 Climate Change

The 2014 Assessment Synthesis Report from the Working Groups on the IPCC concluded climate change is unequivocal (IPCC 2014). The report concludes oceans have warmed, with ocean warming the greatest near the surface (e.g., the upper 75 m [246.1 ft] have warmed by 0.11°C per decade over the period 1971 through 2010) (IPCC 2014). The Atlantic Ocean appears to be warming faster than all other ocean basins except perhaps the southern oceans (Cheng et al. 2017). In the western North Atlantic Ocean, surface temperatures have been unusually warm in recent years (Blunden and Arndt 2016). A study by Polyakov et al. (2009), suggests that the North Atlantic Ocean overall has been experiencing a general warming trend over the last 80 years of 0.031±0.0006°C per decade in the upper 2,000 m (6,561.7 ft) of the ocean. The Fourth National Climate Assessment confirmed that the Atlantic and Gulf coasts in particular are facing above-average risks to ocean and coastal infrastructure (U.S. Global Change Research Program 2018). Also highlighted were rising water temperatures, ocean acidification, retreating arctic sea ice, sea level rise, high tide flooding, coastal erosion, higher storm surge, and heavier precipitation events as key threats to the nation's oceans and coasts (U.S. Global Change Research Program 2018). Climate change is expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, tropical storms, heat waves, and droughts (IPCC 2014; U.S. Global Change Research Program 2018). Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Ocean acidity has increased by 26% since the beginning of the industrial era (IPCC 2014) and this rise has been linked to climate change.

Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure in the action area (Evans and Bjørge 2013; IPCC 2014; Kintisch 2006; Learmonth et al. 2006; MacLeod et al. 2005; McMahon and Hays 2006; Robinson et al. 2009). Marine species' ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Though predicting the precise consequences of climate change on marine species is difficult (Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring (U.S. Global Change Research Program 2018).

Other examples include the McMahon and Hays (2006) study that found increased ocean temperatures are expanding the distribution of leatherback turtles into more northern latitudes in the Atlantic Ocean. On the opposite end of the spectrum, sessile species (e.g., corals and seagrasses) are unable to expand their ranges or leave certain areas to find more suitable habitat, making it more difficult for these species to adapt to warming temperatures.

6 EFFECTS OF THE ACTION

Effects of the action are all consequences to listed species 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 8. 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 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. 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 various effects of the proposed action on loggerhead, green, and Kemp's ridley sea turtles, as well as giant manta ray.

6.1 Effects to Sea Turtles

6.1.1 Dredging Effects

To estimate take from hopper dredging covered under this Opinion, we reviewed the reported take of ESA-listed species from projects in the action area and greater USACE Galveston District provided in Table 6 and compared it to the reported volume of material dredged (effort) by those projects to calculate a hopper dredging CPUE for each species. The reported total volume of material dredged per fiscal year in Table 6 was gathered from multiple sources provided by and/or verified by the USACE, but still may not precisely reflect the volumes dredged. At this time, it is the best available information and will be used for the analysis of take for this Opinion. We assume all sea turtle takes by hopper dredge will be lethal.

Table 5. Sea Turtle Takes Documented in USACE Galveston District Dredging Projects, 2014-2020.

YEAR	PROJECT	CY DREDGED	LOGGERHEAD	GREEN	KEMP'S	UNKNOWN
2020	SABINE-NECHES	5,183,000			1	
2019	FREEPORT	2,164,666		3 (1 ALIVE)		
2019	BRAZOS ISLAND	374,291		8 (1 ALIVE)		
2019	GLAVESTON	2,382,000	1			
2019	CORPUS CHRISTI	6,618,964	9	7 (1 ALIVE)	3	
2018	FREEPORT	1,987,232	4 (1 ALIVE)	2		1
2017	FREEPORT	3,164,978		1 (ALIVE)		
2017	GALVESTON	3,724,491			1	
2017	MATAGORDA	195,000		1		
2016	BRAZOS ISLAND	685,369		2		
2016	CORPUS CHRISTI	846,000	2			
2015	FREEPORT	2,096,850		1		
2014	CORPUS CHRISTI	200,000		2		
2014	FREEPORT	495,000		5		
2014	SABINE-NECHES	4,131,901			1	
2014	BRAZOS ISLAND	304,629		1 (ALIVE)		
TOTAL		**34,554,371**	**16 (1 ALIVE)**	**33 (5 ALIVE)**	**6**	**1**

We first look at all projects in recent years (2014-2020) to estimate CPUE of sea turtles based on the volume of material removed by hopper dredge. Using the total of 34,554,371 CY of dredged material from USACE Galveston District projects 2014-2020 and dividing that by the 56 total turtle takes results in a CPUE of 617,042 CY/turtle. Because there may be geographical, environmental, or sea turtle abundance issues that result in significant differences in sea turtle take at CCSC, we also estimate CPUE for just the 3 projects conducted at CCSC between 2014-2020. The 7,664,964 CY of dredge material removed during USACE Corpus Christi projects from 2014-2020 resulted in 23 total turtle takes, which yields a CPUE of 333,259 CY/turtle.

Of the 46.3 million CY anticipated to be dredged during the proposed project, 32,031,193 CY are expected to be removed via hopper dredge (Table 1). The proposed action indicates that in year 3 the 2,105,041 CY of material may be dredged by either hopper or cutterhead dredge. Given the possibility this material could be removed by a hopper dredge, we will be conservative for the purposes of this analysis and assume this volume of material will be removed by hopper dredge.

We use the previously calculated CPUEs to extrapolate out for the proposed dredging of 32,031,193 CY of material at CCSC. Based on the CPUE estimated from all Galveston District projects between 2014-2020, we estimate 52 sea turtle takes will occur during the proposed action (32,031,193 CY / 617,042 CY/turtle = 52 total turtle takes). And when using the more conservative Corpus Christi projects between 2014-2020, we estimate 96 sea turtle takes will occur during the proposed action (32,031,193 CY / 333,259 CY/turtle = 96 total turtle takes).

In order to calculate species-specific take, we reviewed STSSN data for Texas strandings over the past 5 years (i.e., 2017-2021) to get insight into the relative abundance of sea turtle species in the action area, and to be sure it does not diverge from the species distribution in the USACE Texas dredge take data (Table 6). We use this relative abundance to calculate species-specific take from total anticipated takes calculated above. As presented in Table 7, we expect take to consist of green (75.30%), loggerhead (12.30%), and Kemp's ridley (11.37%) sea turtles. As discussed in Section 3.2, we do not expect take of hawksbill or leatherback sea turtles due to habitat preferences and other rationale. These conclusions are supported by the general rarity of these species in available Texas strandings data, as presented in Table 7.

Table 6. Texas Sea Turtle Strandings (Traditional), 2017-2021 (STSSN Data). We believe the single olive ridley stranding to be an anomalous outlier, as the species is rarely found in the Gulf of Mexico.

SPECIES	NUMBER	PERCENTAGE
GREEN	3,954	75.30
LOGGERHEAD	646	12.30
KEMP'S RIDLEY	597	11.37
HAWKSBILL	43	0.82
LEATHERBACK	10	0.19
OLIVE RIDLEY	1	0.02
TOTAL	**5,251**	**100**

Therefore, erring on the side of the species and using the more conservative CCSC-specific take estimate of 96 total sea turtle takes that we calculated would occur during proposed hopper dredging activities, we further estimate this would consist of 72 green sea turtles (96 * 0.7530 = 72.29; NA and SA DPSs), 12 loggerhead sea turtles (96 * 0.1230 = 11.81), and 11 Kemp's ridley sea turtles (96 * 0.1137 = 10.92). The sum of these species-specific estimates is different than the total sea turtle take estimated previously (i.e., 95 versus 96) due to rounding. So, while we would expect this additional turtle would likely be a green sea turtle based on relative abundance as represented in STSSN and USACE Texas dredge take data (Table 6) we will allocate and add the 1 remaining sea turtle take to any of the 3 affected sea turtle species (i.e., green, loggerhead, or Kemp's ridley).

6.1.2 Relocation Trawling Effects

We have limited information on the quantitative effects of relocation trawling associated with USACE dredging activities to listed species. Because hopper dredging occurs during various times of the year, we would expect associated relocation trawling take (i.e., CPUE) to vary greatly due to seasonality and differences in sea turtle abundance. Additionally, relocation trawling effort varies as well, as it is not always required and sometimes only implemented upon a project trigger (e.g., 2 sea turtle takes within 24 hours). Some past projects have used a single relocation trawler when required, and some projects have employed 2 relocation trawlers concurrently for better channel coverage. The USACE used to produce annual reports as required under the terms and conditions of the GRBO, but they no longer produce those annual reports (J. Hudson, USACE, pers. comm., September 26, 2022). Therefore, we present some historical information included in annual reports available on ODESS, as well as some recent data provided by USACE Galveston District to gain insight into relocation trawling activities and the potential extent of effects on listed species.

During Fiscal Year (FY) 2007, 4 maintenance-dredging projects were conducted by hopper dredges in the Galveston District, during which 4,583,566 CY of sediments were excavated. Where implemented, relocation trawling was performed on a 24-hour daily basis during dredging operations. Two trawlers worked concurrently to provide better channel coverage during dredging at BIH ,while one trawler was used at CCSP. During the course of 2,511 trawls, 102 turtles were relocated consisting of 65 green, 25 Kemp's ridley, and 12 loggerhead sea turtles; this total also includes 2 recaptures (Table 8).

Table 7. FY 2007 Galveston District Relocation Trawler Effort Associated with Dredging Projects (ODESS data).

PROJECT	NUMBER OF TOWS	NUMBER OF TURTLES	CPUE
BIH	996	65	0.0653
CCSP	1,515	37	0.0244
TOTALS	2,511	102	0.0406

During FY 2008, 6 maintenance-dredging projects were conducted by hopper Dredges in the Galveston District, during which 8,104,240 CY of sediments were excavated. Where implemented, relocation trawling was performed on a 24-hour daily basis during dredging operations. Two trawlers worked concurrently to provide better channel coverage during dredging at both BIH projects, while one trawler was used at Freeport. During the course of

2,145 trawls, 17 turtles were relocated; this includes 13 loggerhead, 3 Kemp's ridley, and 1 green sea turtle; this total also includes 3 recaptures (Table 9).

Table 8. FY 2008 Galveston District Relocation Trawler Effort Associated with Dredging Projects (ODESS data).

PROJECT	NUMBER OF TOWS	NUMBER OF TURTLES	CPUE
FREEPORT HARBOR	430	0	0.000
BIH – JETTY CHANNEL	1,304	14	0.0107
BIH – ENTRANCE CHANNEL	411	3	0.0073
TOTALS	2,145	17	0.0079

During FY 2009, 5 maintenance-dredging projects were conducted by hopper dredges in the Galveston District, during which 16,078,665 CY of sediments were excavated. Relocation trawling was only implemented during one project, where 2 trawlers worked concurrently on a 24-hour daily basis to provide better channel coverage during dredging. Over the course of 820 trawls, 1 loggerhead and 1 green sea turtle were relocated, corresponding to a combined CPUE of 0.002 turtles/tow.

In contrast, in FY 2013 and FY 2014 in the USACE Galveston District there were 7 and 5 maintenance dredging projects that removed 3,462,215 CY and 9,308,101 CY of material, respectively, but there were no relocation trawling activities. This helps to demonstrate the irregular use of relocation trawling based on a 2-turtle take trigger during USACE Galveston District projects and, moreover, the difficulty in estimating the effects of this action. This is reinforced by the highly variable CPUE rates presented in Table 10 below.

Recent data from a 2019 CCSC Improvement Project documented hopper dredges removed a total of 6,618,964 CY of material from April 2019 through February 2020 (ODESS data), with relocation trawling implemented from late June through mid-August 2019. During the relocation trawler effort, 2 trawlers worked concurrently to effectively sweep the area. A total of 2,337 tows were conducted that captured 17 loggerhead, 15 Kemp's ridley, and 4 green sea turtles, resulting in a combined CPUE of 0.0154 turtles/tow. For reference, hopper dredging is expected to remove approximately 32,031,193 CY of material over the course of the proposed action.

Table 9. Selected Dredging Activities and Associated Relocation Trawling Effort (ODESS, USACE Galveston District Data).

PROJECT	CY MATERIAL	TOWS	TURTLES	CPUE
FY 2007 GALVESTON DISTRICT TOTAL	4,583,566	2,511	102	0.0406
FY 2008 GALVESTON DISTRICT TOTAL	8,104,240	2,145	17	0.0079
FY 2009 GALVESTON DISTRICT TOTAL	16,078,665	820	2	0.002
2019 CCSC	6,618,964	2,337	36	0.0154

In summary, available information indicates relocation trawling effort varies greatly and, more importantly, the resultant CPUE of captured turtles during relocation trawling effort varies greatly as well.

Estimating the Extent of Effects
We have sporadic data over 15 years in Table 10, however, there are issues with using this data that should be considered. First, relocation trawling occurs over various times of the year where sea turtle abundance may vary. Second, relocation trawling typically is only initiated after 2 takes occur within 24 hours. And third, sea turtle populations have increased in recent years, particularly for green and Kemp's ridley sea turtles. As a result, we believe the 2019 CCSC relocation trawling work is the most applicable and representative to the proposed action. It is important to note this relocation trawling effort was conducted from June through August. The proposed action indicates it will conduct hopper dredging during seasonal windows (December 1 through March 31) when sea turtle abundance is expected to be lower than summer months, but it indicates this would be done "if practicable." Therefore, to be conservative and err on the side of the species, we will assume these seasonal dredge windows are not compulsory, and work could occur outside of these periods, but acknowledge this may result in an overestimate of relocation trawler take.

We use the 2,337 tows conducted during the removal of 6,618,964 CY of material in the 2019 CCSC project, and assume the rate of dredging and relocation trawling (2,337 tows/6,618,964 CY = 0.00035308 tows/CY) will be similar between the 2019 CCSC work and the proposed action. Based on this approach, we estimate the proposed action that will remove 32,031,193 CY of material via hopper dredge could result in 11,309 total relocation trawler tows (0.00035308 tows/CY * 32,031,193 CY = 11,309 tows) over the course of the proposed action.

As noted in Table 10, CPUE of turtles captured during relocation trawling effort varies greatly. To be consistent, however, we will use the 0.0154 CPUE of captured turtles documented in the 2019 CCSC dredging project. This rate is within the range of all documented CPUEs in Table 10 (i.e., 0.002–0.0406), represents recent data directly from the action area, and occurred during warmer summer months when sea turtle abundance may be higher than in colder, winter months. Therefore, given the possibility hopper dredging activities may occur outside of the seasonal dredging windows and the lack of more explicit and accurate relocation trawling effort, we believe the 0.0154 CPUE is conservative and appropriate to use to estimate take from the proposed action. We also acknowledge this could result in an overestimation of effects. Using this CPUE and applying it the 11,309 total estimated relocation trawler tows yields 174 relocation trawler captures of sea turtles over the course of the proposed action. We further examine the species-specific take from this total estimate below.

To calculate the species-specific number of takes from relocation trawling, we again use the relative abundance of sea turtles as represented in STSSN data for Texas (Table 7). As a result, we estimate the total take of 174 relocated sea turtles would consist of 131 green sea turtles (174 * 0.7530 = 131.02), 21 loggerhead sea turtles (174 * 0.1230 = 21.40), and 20 Kemp's ridley sea turtles (174 * 0.1137 = 19.78). The sum of these species-specific estimates is different than the total sea turtle take estimated previously (i.e., 172 versus 174) due to rounding. While we would expect these 2 additional turtles to be green sea turtles based on relative abundance as represented in USACE Texas dredge take and STSSN data (Tables 6 and 7),we will allocate and add the 2 remaining sea turtle take to any of the 3 affected sea turtle species (i.e., green, loggerhead, or Kemp's ridley). Due to the required limited tow times that avoid forced submergence issues, and required PSOs onboard relocation trawlers who implement protective handling and release guidelines for listed species, we expect all takes by relocation trawlers to be non-lethal.

6.2 Effects to Giant Manta Ray

Due to the large size and pelagic habitat preference (i.e., versus benthic habitat preference), we believe giant manta ray will not be adversely affected by hopper dredging itself. We believe the only likely route of adverse effects to giant manta ray is relocation trawling activity, which we discuss below.

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
As noted in Section 6.1.2, we estimate the proposed action will remove 32,031,193 CY of material via hopper dredge, which could result in 11,309 total relocation trawler tows (0.00035308 tows/CY * 32,031,193 CY = 11,309 tows). Data on take of giant manta ray by relocation trawler is unavailable, but we are aware of anecdotal reports of past relocation trawler captures of giant manta ray. We will follow the same protocol used in the 2020 SARBO, which used a CPUE of 0.00019 based on NMFS Northeast Fisheries Observer Program data from 2001-2015 (NMFS 2020). This results in an estimated take of 2 takes of giant manta ray (11,309 tows * 0.00019 CPUE = 2.15) by relocation trawlers during the course of the proposed action.

We expect the limited tow times, required observers monitoring all relocation trawler tows, and relocation trawler crews following the required best handling and release practices will significantly minimize the risk of post-release mortality associated with relocation trawling activities. Therefore, we anticipate the potential estimated take of 2 giant manta ray by relocation trawlers over the course of the proposed action will be non-lethal.

6.3 Summary

We believe the proposed action will have lethal and non-lethal effects on green, loggerhead, and Kemp's ridley sea turtles from hopper dredging activities and relocation trawling effort, respectively, as well as non-lethal effects on giant manta ray from relocation trawling effort. We quantify the take of those species in Table 11 below. Due to rounding when calculating our estimates, an additional 1 sea turtle may be taken during hopper dredging activities, and could be attributed to any of the 3 affected species. Similarly, an additional 2 sea turtles may be taken during relocation trawling, and could be attributed to any of the 3 affected species. Therefore, the number in parenthesis indicates the potential greatest amount of take of the species by the indicated activity.

Table 10. Summary of Expected Take Resulting From the Proposed Action.

ACTIVITY	SPECIES TAKE			
	GREEN SEA TURTLE	LOGGERHEAD SEA TURTLE	KEMP'S RIDLEY SEA TURTLE	GIANT MANTA RAY
HOPPER DREDGING (LETHAL)	72 (73)	12 (13)	11 (12)	-
RELOCATION TRAWLING (NON-LETHAL)	131 (133)	21 (23)	20 (22)	2

7 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 and giant manta ray, 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 Gulf of Mexico are known to adversely affect sea turtles and giant manta ray. The past and present impacts of these activities have been discussed in Section 5 (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 and giant manta ray covered by this Opinion.

As discussed in Section 4 and, more specifically, Section 5.4, we generally expect climate change may affect sea turtles and giant manta ray, 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 do not expect to observe any climate change effects during the time frame of the proposed action (i.e., 5 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 5 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. For the remaining activities and associated threats identified in Section 5, 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 action area in the future (i.e., over the 5-year time frame of the proposed action) would be similar to those in the past and, therefore, are reflected in the anticipated trends described in Sections 4 and 5.

8 JEOPARDY ANALYSIS

To "jeopardize the continued existence of" of a species means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and the recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Thus, in making this determination for each species, we must look at whether the proposed actions directly or indirectly reduce the reproduction, numbers, or distribution of a listed species. If there is a reduction in 1 or more of these elements, we evaluate whether it would be expected to cause an appreciable reduction in the likelihood of both the survival and the recovery of the species.

The NMFS and USFWS's ESA Section 7 Handbook (USFWS and NMFS 1998) defines survival and recovery, as they apply to the ESA's jeopardy standard. Survival means "the species' persistence…beyond the conditions leading to its endangerment, with sufficient resilience to allow recovery from endangerment." The Handbook further explains that 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. Per the Handbook and the ESA regulations at 50 CFR 402.02, recovery means the "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 or threats to the species are removed so self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

The 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 green (NA and SA DPSs), loggerhead (NWA DPS), and Kemp's ridley sea turtles, as well as giant manta ray. In Section 6, 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 4), the Environmental Baseline (Section 5), and the Cumulative Effects (Section 7), are likely to jeopardize their continued existence in the wild.

The status of each listed species or DPS likely to be adversely affected by the proposed action is reviewed in Section 4. 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 green (NA and SA DPSs), loggerhead (NWA DPS), and Kemp's ridley sea turtles, as well as giant manta ray, to the effects of the action.

8.1 Green Sea Turtle

As noted in Section 4, 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, which are calculated in Table 12 below. Due to rounding when calculating our estimates, an additional 1 sea turtle may be taken during hopper dredging activities, and could be attributed to any of the 3 affected species. Similarly, an additional 2 sea turtles may be taken during relocation trawling, and could be attributed to any of the 3 affected species. Therefore, the number in parenthesis indicates the potential greatest amount of take of the species by the indicated activity.

Table 11. Total Green Sea Turtle Take Estimates by DPS.

ACTIVITY	GREEN SEA TURTLE		
	TOTAL	NA DPS (96%)	SA DPS (4%)
HOPPER DREDGING (LETHAL)	72 (73)	69 (70)	3 (4)
RELOCATION TRAWLING (NON-LETHAL)	131 (133)	126 (128)	5 (7)

8.1.1 Green Sea Turtle NA DPS

Survival

We estimate hopper dredging will result in the lethal take of up to 70 green sea turtles and relocation trawling effort will result in the non-lethal capture of up to 128 green sea turtles from the NA DPS. The non-lethal capture of up to 128 green sea turtles from the NA DPS over the 3 years of anticipated hopper dredging activity (see Table 1) is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Non-lethal captures will not result in a reduction in numbers of the species, as we anticipate these non-lethal captures to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures would be released within the same general area where caught (i.e., within 10 nm), we anticipate no change in the distribution of NA DPS green sea turtles. The potential mortality of up to 70 green sea turtles from the NA DPS over the course of the proposed action would reduce the number of NA DPS green sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. These moralities would also result in a 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. While these mortalities are anticipated to occur within the action area, 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 4 (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 5 (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 5.4. Section 7 (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 fisheries, effects from other federal actions, and the potential effects of climate change, all of which we discussed in detail in the preceding sections of this Opinion. It is important to note that virtually all of the effects discussed 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 4 of this Opinion, nesting has increased substantially over the last 20 years peaking in 2019 with 40,911 nests at the index beaches in Florida. Nesting dropped again in 2020 as expected with the regular biennial fluctuation, but not as much of a drop as in the fast fluctuations, and then rebounded a bit in 2021, as the extreme high/low pattern we've seen in the past appears to be changing to some degree.

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, up to 70 green sea turtles over 3 years represents a very small fraction (<0.2% annually) of the overall NA DPS female nesting population estimated by Seminoff et al. (2015).

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 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 the mortality of up to 70 green sea turtles over the 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.

Recovery

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 6 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 5-year period of statewide nesting beach survey data, there were 53,102 in 2017, 4,546 in 2018, 53,011 in 2019, 26,656 in 2020, and 32,680 in 2021 (see https://myfwc.com/research/wildlife/sea-turtles/nesting/monitoring/). 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.

The potential lethal take of up to 70 green sea turtles from the NA DPS as a result of the proposed action considered in this Opinion is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects.. Thus, the proposed action will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of NA DPS green sea turtles' recovery in the wild.

<u>*Conclusion*</u>
The combined lethal and non-lethal take of green sea turtles from the NA DPS associated with the proposed action 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.

8.1.2 Green Sea Turtle SA DPS

Survival
We estimate hopper dredging will result in the lethal take of up to 4 green sea turtles and relocation trawling will result in the non-lethal capture of up to 7 green sea turtles from the SA DPS. The non-lethal capture of up to 7 green sea turtles from the SA DPS over the course of the project (i.e., 3 years of hopper dredging activity; see Table 1) is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individual suffering non-lethal injuries or stresses is expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures would be released within the same general area where caught (i.e., within 10 nm), we anticipate no change in the distribution of SA DPS green sea turtles. The potential mortality of up to 4 green sea turtles from the SA DPS over the course of the proposed action would reduce the number of SA DPS green sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. 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. While these mortalities are anticipated to occur within 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 4 (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 5 (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 5.4. Section 7 (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 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 Opinion. It is important to note that virtually all of the effects discussed 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 up to 4 sea turtles from this DPS 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. But given the short time period of the proposed action (i.e., 3 years of potential effects from hopper dredging and relocation trawling), we do not expect the effects of climate change will present a risk to the SA DPS green sea turtle population.

After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the SA DPS of green sea turtle in the wild.

Recovery
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 6 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 mortality of up to 4 green sea turtles from the SA DPS 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 non-lethal capture of up to 7 green sea turtles from the SA DPS to have any detectable influence on the recovery objectives above and will not result in an appreciable reduction in the likelihood of the SA DPS of green sea turtles' recovery in the wild.

Conclusion
The potential lethal take of up to 4 green sea turtles from the SA DPS as a result of the proposed action considered in this Opinion is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects.

8.2 Loggerhead Sea Turtle (NWA DPS)

Survival

We estimate that hopper dredging will result in the lethal take of up to 13 loggerhead sea turtles and relocation trawling will result in the non-lethal capture of up to 23 loggerhead turtles from the NWA DPS. The non-lethal capture of up to 23 loggerhead sea turtles (NWA DPS) over the course of the project (i.e., 3 years of hopper dredging activity; see Table 1) is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Individuals suffering non-lethal injuries or stresses are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures would be released within the same general area where caught (i.e., within 10 nm), we anticipate no change in the distribution of NWA DPS loggerhead sea turtles.

The potential mortality of up to 13 loggerhead sea turtles from the NWA DPS over the course of the proposed action would reduce the number of NWA loggerhead sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same.. 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 female loggerhead sea turtle can lay approximately 4 clutches of eggs every 3-4 years, with 100-126 eggs per clutch. Thus, the loss of adult females could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. However, the potential lethal take during any consecutive 3-year period is expected to occur in a small, discrete area and loggerhead sea turtle generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In Section 4 (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 5 (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 5.4. Section 7 (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 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 Opinion. It is important to note that virtually all of the effects discussed 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.

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 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; 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. 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 1,000,000. NMFS (2011) 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 (2011) underestimates the total population of loggerheads since it did not include Florida’s east coast south of Cape Canaveral or the Gulf of Mexico, which are areas where large numbers of loggerheads can also be found. In other words, it provides an estimate of a subset of the entire population. These numbers were derived prior to additional years of increased nesting.

Florida accounts for more than 90% of U.S. loggerhead nesting. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but non-significant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability from 2012-2016 resulting in widening confidence intervals. Nesting at the core index beaches declined in 2017 to 48,033, and rose again each year through 2020, reaching 53,443 nests before dipping back to 49,100 in 2021. 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 some 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 13 individuals over the duration of the proposed action (i.e., 3 years of hopper dredging activity; see Table 1), or an annual average of approximately 4 loggerhead sea turtles. These removed individuals represent approximately 0.00105% annually on the low end of the NMFS (2011) 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 up to 13 individuals 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.

After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe that the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the NWA DPS of loggerhead sea turtle in the wild.

Recovery

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:

- 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.
- 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.

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 up to 13 loggerhead sea turtles 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 and will not result in an appreciable reduction in the likelihood of NWA DPS of loggerhead sea turtles' recovery in the wild.

While the take of neritic juveniles may occur during the proposed action, relocation trawling measures are in place to avoid or minimize lethal take by hopper dredges. For this reason, we do not believe the proposed action will impede achieving this recovery objective and will not result in an appreciable reduction in the likelihood of NWA DPS of loggerhead sea turtles' recovery in the wild.

Conclusion
The potential mortality of up to 13 loggerhead sea turtles from the NWA DPS will result in a reduction in numbers and reproduction 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 4 (Status of the Species), 5 (Environmental Baseline), and 7 (Cumulative Effects) discussed in this Opinion. Similarly, we do not expect the non-lethal capture of up to 23

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 is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects.

8.3 Kemp's Ridley Sea Turtle

Survival

We estimate hopper dredging will result in the lethal take of up to 12 Kemp's ridley sea turtles, and relocation trawling will result in the non-lethal capture of up to 22 Kemp's ridley sea turtles. The non-lethal capture of up to 22 Kemp's ridley sea turtles over the course of the project (i.e., 3 years of hopper dredging activity; see Table 1) is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individual suffering non-lethal injuries or stresses are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures would be released within the same general area where caught (i.e., within 10 nm), we anticipate no change in the distribution of Kemp's ridley sea turtles. The mortality of up to 12 Kemp's ridley sea turtles over the course of the proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed actions, assuming all other variables remained the same. 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.

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 4 (Status of Species), we presented the status of the species, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In Section 5 (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 5.4. Section 7 (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 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 Opinion. It is important to note that virtually all of the effects discussed 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 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 data, 2019). Nesting numbers rebounded in 2020 (18,068 nests) and 2021 (17,671 nests) (CONAMP data, 2021). 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 (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, rebounding to 262 nests in 2020, and back to 195 nests in 2021 (NPS data).

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 1 or 2 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 6 distinct haplotypes, with one found at both Padre Island and Rancho Nuevo (Dutton et al. 2006).

The proposed action could remove up to 12 individuals over the duration of the proposed action (i.e., 3 years of hopper dredging activity as documented in Table 1), or an annual average of approximately 4 Kemp's ridley sea turtles. These removed individuals represent approximately 0.018% annually of the sexually-mature population estimated in Wibbels and Bevan (2019). While the loss of up to 12 individuals 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.

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 with all the adverse effects included in the baseline. The loss of 12 Kemp's ridleys over the course of the proposed action is not expected to change the trend in nesting, the distribution of, or the reproduction of Kemp's ridley sea turtles. Therefore, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

Recovery

As to whether the dredging will appreciably reduce the species' likelihood of recovery, the recovery plan for the Kemp's ridley sea turtle (NMFS et al. 2011) lists the following relevant recovery objective:

- 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.

With respect to the demographic recovery objective, the nesting numbers in the most recent 3 years indicate there were 11,090 nests in 2019, 18,068 in 2020, and 17,671 in 2021 on the main nesting beaches in Mexico. Based on 2.5 clutches/female/season, these numbers represent approximately 4,436 (2019), 7,227 (2020), and 7,068 (2021) 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. Non-lethal captures of these sea turtles would not affect the adult female nesting population or number of nests per nesting season.

Conclusion
The potential lethal take of up to 12 Kemp's ridley sea turtles as a result of the proposed action considered in this Opinion is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects.

8.4 Giant Manta Ray

We estimate relocation trawling will result in the non-lethal capture of up to 2 giant manta ray.

Survival
The non-lethal capture of up to 2 giant manta ray over the course of the project (i.e., 3 years of hopper dredging activity; see Table 1) is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals captured are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures would be released within the same general area where caught (i.e., within 10 nm), we anticipate no change in the distribution of giant manta ray.

Recovery
A recovery plan for giant manta ray has not yet been developed; however, we published a recovery outline for the species (NMFS 2019). The recovery outline serves as an interim guidance to direct recovery efforts for giant manta ray. The recovery outline identifies two primary interim goals:

- Stabilize population trends through reduction of threats, such that the species is no longer declining throughout a significant portion of its range; and
- Gather additional information through research and monitoring on the species' current distribution and abundance, movement and habitat use of adult and juveniles, mortality rates in commercial fisheries (including at-vessel and PRM), and other potential threats that may contribute to the species' decline.

The major threats affecting the giant manta ray were summarized in the final listing rule (83 FR 2619, Publication Date January 22, 2018). The most significant threats to the giant manta ray are overutilization by foreign commercial and artisanal fisheries in the Indo-Pacific and Eastern Pacific and inadequate regulatory mechanisms in foreign nations to protect this species from the heavy fishing pressure and related mortality in these waters outside of U.S. jurisdiction. Other threats that potentially contribute to long-term risk of the species include: (micro)plastic ingestion rates, increased parasitic loads as a result of climate change effects, and potential disruption of important life history functions as a result of increased tourism. However, due to the significant data gaps, the likelihood and impact of these threats on the status of the species is highly uncertain. Recreational fishing interactions are not considered a major threat to this species and we do not believe the proposed action will appreciably reduce the recovery of giant manta ray, by significantly exacerbating effects of any of the major threats identified in the final listing rule.

The individuals suffering non-lethal capture are expected to fully recover such that no reductions in reproduction or numbers of giant manta rays are anticipated. The non-lethal capture will occur at in a discrete location and the action area encompasses only a portion of the overall range or distribution of giant manta rays. Any incidentally caught animal would be released within the general area where caught and no change in the distribution of giant manta rays would be anticipated. Therefore, the non-lethal capture of giant manta rays associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of recovery of the giant manta rays in the wild.

Conclusion

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 non-lethal capture of 2 giant manta rays as a result of the proposed action considered in this Opinion is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects.

9 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the RPMs and the terms and conditions of the ITS of the Opinion.

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.

The take of the giant manta ray by the proposed action is not prohibited, as no Section 4(d) Rule for the species has been promulgated. However, a 9th Circuit Court case held that non-prohibited incidental take must be included in the ITS (*CBD v. Salazar*, 695 F.3d 893 [9th Cir. 2012]). Though the *Salazar* case is not a binding precedent for this action outside of the 9th Circuit, we find the reasoning persuasive and is following the case out of an abundance of caution and anticipates the ruling will be more broadly followed in future cases. Providing an exemption from Section 9 liability is not the only important purpose of specifying take in an ITS. Specifying incidental take ensures we have a metric against which we can measure whether or not reinitiation of consultation is required. It also ensures that we identify RPMs we believe are necessary or appropriate to minimize the impact of such incidental take.

9.1 Anticipated Incidental Take

As discussed in Section 6, we anticipate the proposed action will result in the take of green (NA and SA DPSs), loggerhead (NWA DPS), and Kemp's ridley sea turtles, as well as giant manta ray, as summarized in Table 13. Due to rounding and other issues when calculating our estimates, an additional 1 sea turtle may be taken during hopper dredging activities, and could be attributed to any of the 3 affected species. An additional 2 sea turtles may be taken during relocation trawling, and could be attributed to any of the 3 affected species. Therefore, the number in parenthesis indicates the potential greatest amount of take of the species by the indicated activity.

Table 12. Summary of Expected Take Resulting From the Proposed Action.

ACTIVITY	SPECIES TAKE			
	GREEN SEA TURTLE	LOGGERHEAD SEA TURTLE	KEMP'S RIDLEY SEA TURTLE	GIANT MANTA RAY
HOPPER DREDGING (LETHAL)	72 (73)	12 (13)	11 (12)	0
RELOCATION TRAWLING (NON-LETHAL)	131 (133)	21 (23)	20 (22)	2

9.2 Effect(s) of the Take

We have determined that the anticipated take specified in Section 9.1 is not likely to jeopardize the continued existence of Kemp's ridley, green (NA and SA DPSs), and loggerhead (NWA DPS) sea turtles, as well as giant manta ray, as a result of the proposed action.

9.3 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires us to issue to any federal 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. The ITS must specify the RPMs necessary to minimize the impacts of the incidental taking from the proposed action on the species, and Terms and Conditions to implement those measures. Per Section 7(o)(2), and incidental taking that complies with the specified terms and conditions is not considered to be prohibited taking of the species concerned.

The RPMs and terms and conditions are required to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species 50 CFR 402.14 (i)(1)(ii) and (iv). These measures and terms and conditions must be implemented by USACE for the protection of Section 7(o)(2) to apply. The USACE has a continuing duty to ensure compliance with the reasonable and prudent measures and terms and conditions included in this ITS. If it fails to adhere to the terms and conditions of the ITS through enforceable terms, or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse. To monitor the impact of the incidental take, the USACE must report the progress of the action and its impact on the species to us, as specified in the ITS (per 50 CFR 402.14(i)(3)).

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: Avoidance/Mitigation of Project-Related Effects

The USACE and its contractors must have measures in place to avoid and/or minimize interactions with any protected species resulting from the proposed action, as appropriate.

RPM 2: Handling

USACE must ensure observers handle sea turtles and giant manta ray in a manner that prevents injury and helps ensure survivability upon release.

RPM 3: Reporting

USACE must notify local STSSN of all activities and report to us any dredge takes that occur during the proposed action.

9.4 Terms and Conditions

To be exempt from take prohibitions established by Section 9 of the ESA, USACE 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:

1. From March 15 through October 1, sea turtle nesting and emergence season, all lighting aboard all dredges and support vessels operating within 3 nm of sea turtle nesting beaches would be limited to the minimal lighting necessary to comply with USCG and OSHA requirements. Non-essential lighting must be minimized through reduction, shielding, lowering, and appropriate placement.
2. Any PSO contracted by USACE or PCCA must be NMFS-approved.
3. Relocation trawling must be undertaken by a NMFS-approved PSO retained by the PCCA if hopper dredging activities result in either (a) 2 or more lethal sea turtle takes occur in a 24-hour period or, (b) more than 4 lethal sea turtle takes occur during the proposed action.
4. A state-of-the-art rigid deflector draghead must be used on hopper dredges at all times of the year. Dredging pumps must be disengaged by the operator when the dragheads are not firmly on the bottom as indicated by sensors to prevent impingement or entrainment of sea turtles within the water column.
5. NMFS-approved PSOs must be aboard the hopper dredges or disposal barge during material placement. Operations shall cease if an ESA-listed species is observed within 150 feet of operations. Activities shall not resume until the protected species has departed the project area of its own volition (e.g., species was observed departing or 20 minutes have passed since the animal was last seen in the area).

The following term and condition implement RPM 2:

1. Proper handling of any protected species incidentally caught during relocation trawling operations is essential to increase the likelihood of its survival. For giant manta ray and sea turtles, observers must use the safe handling and release guidelines provided in Appendix 2.

The following terms and conditions implement RPM 3:

1. NMFS-approved PSOs must be aboard the hopper dredges to monitor the hopper bin, screening, and dragheads for sea turtles and their remains. Observer coverage sufficient for 100% monitoring (i.e., 2 observers) of hopper dredging operations must be implemented.
2. Observer reports of incidental take by hopper dredges must be submitted by email (takereport.nmfsser@noaa.gov) to us by onboard PSOs within 24 hours of any observed sea turtle take. Reports must contain information on location, start-up and completion dates, cubic yards of material dredged, problems encountered,

incidental takes, and sightings of protected species, mitigative actions taken, screening type, and daily water temperatures.

3. An end-of-project summary report of the project, including dredge takes by species and relocation trawler effort, must be posted to the USACE ODESS website within 30 working days of completion of the proposed action.
4. USACE will insure PCCA or its representative that it must notify the Texas STSSN representative of start-up and completion of dredging and relocation trawling operations. The STSSN must be notified of any turtle strandings in the project area that may bear the signs of interaction with a dredge. Stranded sea turtles would be reported to the Texas sea turtle hotline (1-866-TURTLE5 or 1-866-887-8535).

10 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 carried out by USACE:

1. We recommend the USACE upload historical dredging reports to ODESS and maintain the repository to aid future Section 7 consultations on dredging projects.

2. We recommend the USACE require all personnel to report giant manta ray sightings to the giant manta ray recovery coordinator at SERO PRD. Giant manta ray’s observations should be photographed and include the latitude/longitude, date, and environmental conditions at the time of the sighting.

11 REINITIATION OF CONSULTATION

This concludes formal consultation on the proposed action. As provided in 50 CFR 402.16, reinitiation of formal consultation is required and shall be requested by the USACE or by the Service, 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, the USACE must immediately request reinitiation of formal consultation and project activities may only resume if the USACE establishes that such continuation will not violate Sections 7(a)(2) and 7(d) of the ESA.

12 LITERATURE CITED

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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 non-lethal)	7 (lethal or non-lethal)	-	-	-
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 non-lethal)	4 (lethal or non-lethal)	-	-	-
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 non-lethal)	1 (lethal or non-lethal)	-	-	-

[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 non-lethal)	-	12 (lethal or non-lethal)	12 (lethal or non-lethal)
Gulf of Mexico/South Atlantic Spiny Lobster (August 27, 2009)	3 years	3 (lethal or non-lethal)	1 (lethal or non-lethal)		3 (lethal or non-lethal)	1 (lethal or non-lethal)
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
Southeast Shrimp Fisheries (April 26, 2021)	5 years	72,670; 2,150 lethal	130; 5 lethal	84,495; 8,505 lethal	21,214; 1,700 lethal	170; 5 lethal

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

Fishery	ITS Period	Giant Manta Ray
HMS, Excluding Pelagic Longline (January 10, 2020)	3 years	9 non-lethal
Southeast Shrimp Fisheries (April 26, 2021)	5 years	8,390 non-lethal

APPENDIX 2 SPECIES HANDLING AND RELEASE GUIDELINES

NOAA FISHERIES

Sea Turtle Handling and Resuscitation Requirements

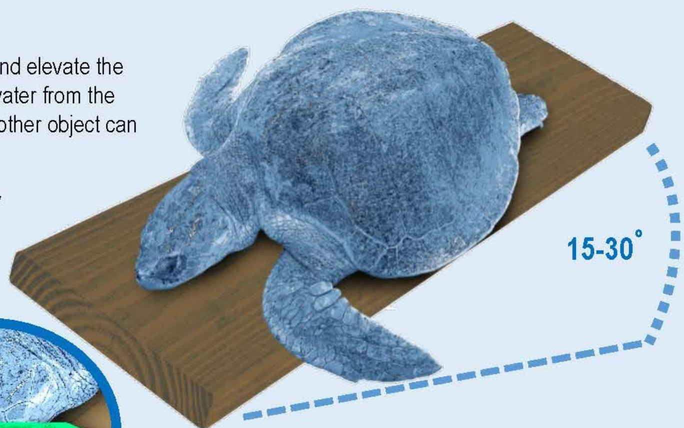
Per federal regulations at 50 CFR 223.206(d)(1):

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.

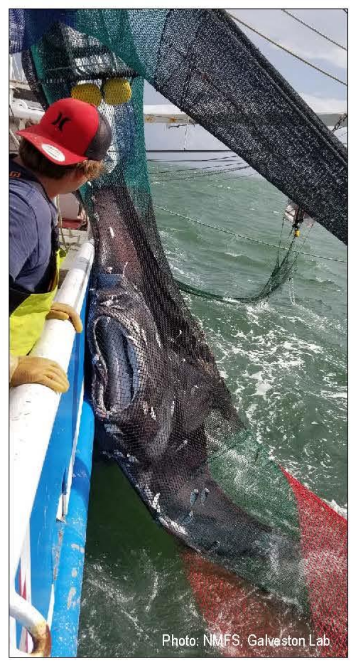


Photo: NMFS, Galveston Lab