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Mr. Philip Hegji, South Mississippi Section  
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Dear Mr. Hegji:

The enclosed Biological Opinion (“Opinion”) responds to your request for consultation with us, the National Marine Fisheries Service (NMFS), pursuant to Section 7 of the Endangered Species Act (ESA) for the following action.

The Opinion considers the effects of the U.S. Army Corps of Engineers (USACE) Mobile District permitting the placement of high-relief and low-relief materials of opportunity located within 7 of 14 existing permitted Fish Haven (FH) artificial reef sites in the Mississippi Sound and the Gulf of Mexico over a 5-year period. We base this Opinion on project-specific information provided in the consultation package as well as NMFS’s review of published literature. This Opinion analyzes the potential for the project to affect the following species: green sea turtle (North Atlantic and South Atlantic distinct population segments [DPSs]), hawksbill sea turtle, Kemp’s ridley sea turtle, leatherback sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), Gulf sturgeon, oceanic whitetip shark, giant manta ray, and Bryde’s whale. This Opinion also analyzes the potential for the project to have an effect on Gulf sturgeon critical habitat (Lake Pontchartrain – Mississippi Sound – Unit 8). Pursuant to 50 CFR 402.14(i)(5), any taking which is subject to an incidental take statement (ITS), and which is in compliance with the terms and conditions (T&C) associated with the ITS, is not a prohibited taking under the ESA.

Please direct any questions to Daniel Owen, Consultation Biologist, by phone at (727) 209-5961, or by email at [Daniel.Owen@noaa.gov](mailto:Daniel.Owen@noaa.gov).

Sincerely,

Andrew J. Strelcheck  
Acting Regional Administrator

Enclosures:  
Biological Opinion

File: 1514-22 f.6



**Endangered Species Act - Section 7 Consultation  
Biological Opinion**

**Action Agency:** U.S. Army Corps of Engineers, Mobile District

**Applicant:** Mississippi Gulf Fishing Banks, Inc.

Permit Numbers SAM-2018-1011(FH-1); SAM-2018-1012(FH-2); SAM-2018-1013(FH-3); SAM-2018-1018(FH-8) ; SAM-2018-1019(FH-9/11); and SAM-2018-1020(FH-10).

**Activity:** Mississippi Gulf Fishing Banks Artificial Reef

**Consulting Agency:** National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida

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**Approved by:**

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

**Date Issued:**

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### Acronyms and Abbreviations

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CFR	Code of Federal Regulations
CPUE	catch per unit effort
DDT	dichlorodiphenyltrichloroethane
DO	dissolved oxygen
DoD	Department of Defense
DPS	Distinct Population Segment
DWH	Deepwater Horizon
ESA	Endangered Species Act
FERC	Federal Energy Regulatory Commission
FH	Fish Haven
FMP	Fishery Management Plan
FP	fibropapillomatosis
FR	Federal Register
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Florida Fish and Wildlife Research Institute
GADNR	Georgia Department of Natural Resources
GSCH	Gulf sturgeon critical habitat
ITS	Incidental Take Statement
LAA	Likely to Adversely Affect
MDMR	Mississippi Department of Marine Resources
MGFB	Mississippi Gulf Fishing Banks
NWR	National Wildlife Refuge
NA DPS	North Atlantic DPS of green sea turtle
NCWRC	North Carolina Wildlife Resources Commission
NE	No Effect
NGMRU	Northern Gulf of Mexico Recovery Unit
NLAA	Not Likely to Adversely Affect
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration

NPS	National Park Service
NRU	Northern Recovery Unit
NWA DPS	Northwest Atlantic distinct population segment
NWALWG	Northwest Atlantic Leatherback Working Group
Opinion	Biological Opinion
OSAT	Operational Science Advisory Team
PCB(s)	polychlorinated biphenyl(s)
PCE(s)	Primary Constituent Element(s)
PDC	project design criteria
PFRU	Peninsular Florida Recovery Unit
RPMs	reasonable and prudent measures
SA DPS	South Atlantic DPS of green sea turtle
SCDNR	South Carolina Department of Natural Resources
SCL	straight carapace length
SOM(s)	submerged oil mats
STSSN	Sea Turtle Stranding and Salvage Network
T&Cs	terms and conditions
TED(s)	Turtle Excluder Device(s)
TEWG	Turtle Expert Working Group
U.S.	United States of America
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service

### **Units of Measurement**

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°C	degree Celsius
CCL	curved carapace length
cm	centimeter(s)
CPUE	catch per unit effort
ft	foot/feet
g	gram(s)
in	inch(es)
kg	kilogram(s)
km	kilometer(s)
km <sup>2</sup>	square kilometer(s)
lb	pound(s)
nm	nautical mile(s)
m	meter(s)
mm	millimeter(s)
mi	mile(s)
oz	ounce(s)
rkm	river kilometer(s)
SCL	straight carapace length
yd	yard(s)
yd <sup>3</sup>	cubic yard(s)

## INTRODUCTION

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When the action of a federal agency may affect a protected species or its critical habitat, that agency is required to consult with either National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the protected species that may be affected.

Consultations on most listed marine species and their designated critical habitat are conducted between the action agency and NMFS. Consultations are concluded after NMFS determines the action is not likely to adversely affect listed species or critical habitat or issues a Biological Opinion (“Opinion”) that determines whether a proposed action is likely to jeopardize the continued existence of a federally listed species, or destroy or adversely modify federally designated critical habitat. The Opinion also states the amount or extent of listed species incidental take that may occur and develops nondiscretionary measures that the action agency must take to reduce the effects of said anticipated/authorized take. The Opinion may also recommend discretionary conservation measures. No incidental destruction or adverse modification of critical habitat may be authorized. The issuance of an Opinion detailing NMFS’s findings concludes ESA Section 7 consultation.

This document represents NMFS’s Opinion based on our review of the effects of placing high-relief materials of opportunity (such as vessels, aircrafts, decommissioned oil rigs, bridge spans, metal towers, and other large metal structures) in Fish Haven (FH) sites 1, 2, and 7, and low-relief materials of opportunity (such as solid concrete, rock rubble, reef balls, etc.) in Gulf sturgeon critical habitat (GSCH) in FH sites 8-11, which are 7 of the 14 existing FH artificial reef sites in the Mississippi Sound and the Gulf of Mexico. Work would be conducted over a 5-year period. This Opinion analyzes the potential for the project to affect the following species: loggerhead sea turtle (Northwest Atlantic distinct population segments [DPSs]), green sea turtle (North Atlantic and South Atlantic DPSs), Kemp’s ridley sea turtle, leatherback sea turtle, hawksbill sea turtle, Gulf sturgeon, oceanic whitetip shark, giant manta ray, and Bryde’s whale.<sup>1</sup> This Opinion also analyzes the potential for the project to have an effect on GSCH (Lake Pontchartrain – Mississippi Sound – Unit 8). NMFS has analyzed the information provided by the U.S. Army Corps of Engineers (USACE) and the effects on listed species under our purview in accordance with Section 7 of the ESA of 1973, as amended (16 U.S.C. 1531 et seq.). NMFS bases this Opinion on project information provided by the USACE as well as published literature and the best available scientific and commercial information.

### 1. CONSULTATION HISTORY

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The following is the consultation history for identifier number SERO-2019-03400, Mississippi Gulf Fishing Banks Artificial Reef:

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<sup>1</sup> On August 23, 2021, NMFS announced the revised taxonomy and name of *Balaenoptera edeni* (unnamed subspecies; Bryde’s Whale—Gulf of Mexico subspecies) to the scientific name of *Balaenoptera ricei* and the common name of Rice’s whale, which will become effective on October 22, 2021. 86 Fed. Reg. 47,022 (August 23, 2021).

- On March 14, 2019, NMFS received a request from the USACE for informal consultation on permits to deploy materials of opportunity and materials of design within existing FH sites in the Mississippi Sound and the Gulf of Mexico.
- On April 18, July 9, and November 1, 2019, NMFS sent requests for additional information based on the high relief material and low relief material located in critical habitat and to suggest additional Project Design Criteria (PDCs) to be implemented as part of the proposed action. We received a final response on November 22, 2019, and initiated consultation that day (SERO-2019-03400).
- On May 4, 2021, NMFS requested additional information during our internal review process. NMFS received a response on May 5, 2021.
- On Jul 28, 2021, NMFS issued a draft Biological Opinion for USACE comment. NMFS received comments in response on August 25, 2021.

After reviewing the data, NMFS concluded that work at seven of the locations could be handled as an informal consultation and issued a letter of concurrence (LOC) for those seven FHs (FH-3, FH-4, FH-5, FH-6, FH-12, FH-13, and FH-14) to the USACE on December 12, 2019 (SERO-2019-00045). The remaining sites (FH-1, FH-2, FH-7, FH-8, FH-9, FH-10 and FH-11), which are either high-relief artificial reefs or low-relief artificial reefs in designated critical habitat, require formal consultation.

## **2. DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA**

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### **2.1 Proposed Action**

This Opinion evaluates the effects of placing high-relief and low-relief artificial reef materials within the existing FH artificial reef sites in the Mississippi Sound and the Gulf of Mexico that were not covered by the LOC issued to the USACE on December 12, 2019 (SERO-2019-00045). Specifically, this Opinion evaluates the effects of placing high-relief materials at existing FH sites FH-1, FH-2 and FH-7, as well as the effects of placing low-relief materials at existing FH sites that are located within Gulf sturgeon designated critical habitat (FH-8, FH-9, FH-10 and FH-11). The USACE authorizes this activity pursuant to its permit numbers SAM-2018-01011-23 and 77-PAH. The other seven FHs that were included under the USACE permit and covered by the informal consultation in SERO-2019-00045 (FH-3, FH-4, FH-5, FH-6, FH-12, FH-13, and FH-14) are low-relief artificial reefs that are not located in designated critical habitat. The low-relief artificial reefs (FH-8, FH-9, FH-10 and FH-11) included in this Opinion are the artificial reefs located in Gulf sturgeon critical habitat (Unit 8).

The USACE proposes to permit the Mississippi Gulf Fishing Banks, Inc. (MGFB or the applicant) to enhance existing artificial reefs by placing high-relief materials of opportunity at FH-1, FH-2, and FH-7, and low-relief materials of opportunity located within GSCH at FH-8, FH-9, FH-10 and FH-11, in the Mississippi Sound and the Gulf of Mexico over a 5-year period. MGFB anticipates deploying up to five structures as high-relief reef material in FH-1, up to six structures as high-relief reef material in FH-2, and up to four structures as high-relief reef material in FH-7 over the next five years. Only low-relief material such as concrete rubble and reef balls will be deployed at sites FH-8, FH-9, FH-10 and FH-11, which are located within GSCH.

Low-relief reef material includes materials such as solid concrete, rock rubble, and individual artificial reef modules that present less complicated vertical relief. NMFS considers high-relief, complex artificial reef material to include any vessel, aircraft, decommissioned oil rig, bridge span, metal tower, or similar material that extends 7 feet (ft) or more from the seafloor and that has a footprint greater than 200 square feet (ft<sup>2</sup>) (individually or collectively), excluding prefabricated artificial reef modules.

#### *Artificial Reef Construction and Deployment*

Reef material shall periodically be transported to deployment areas in accordance with the vessel movement Project Design Criteria described below on shallow draft barges that are loaded to conform to the depth of the water at each specific deployment site and within the boundaries of the reef zones. Material will be deployed via clamshell and/or excavator bucket and a minimum clearance of 7 feet (ft) will be maintained at each site from the top of the structure to the water surface. An average rubble pile or culvert deployment is approximately 369 cubic yards (yd<sup>3</sup>) of material. The smallest area covered is approximately 14x14 meters (m) or 2,066 square feet (ft<sup>2</sup>) with a height of 1-2 m. MGFB designee(s) or Mississippi Department of Marine Resources (MDMR) personnel will be onsite to monitor compliance with the terms of the permit. The applicant estimates that the high relief materials of opportunity will require up to five vessels in Fish Haven 1, up to six vessels in Fish Haven 2, and up to four vessels in Fish Haven 7 over the next five years. Reef deployment will take 1 to 2 days per vessel to be deployed as reef material. In-water work will occur during daylight hours only. Personnel will be advised that there are civil and criminal penalties for harming, harassing or killing species covered under the Endangered Species Act of 1973. To further reduce adverse effects to listed species as a result of the proposed action, the following Project Design Criteria (PDCs) will be incorporated as a condition of MGFB's USACE permit:

#### Material:

- 1) Reef material shall consist of clean nontoxic concrete rubble and materials of design (reef balls), and all materials will be consistent with materials listed in Guidelines for Marine Artificial Reef Materials, 2nd edition and the National Artificial Reef Plan (as Amended): Guidelines for Siting, Construction, Development, and Assessment of Artificial Reefs.
- 2) Materials of design such as "reef balls" used for offshore deployments will have a solid base of approximately 6 ft in diameter and a height of 5 ft. The reef ball will have a minimum of 18 openings along its surface. The top of the reef ball will have a minimum of a 42-inch (in) opening. The reef balls will be deployed in a manner that ensures the placement of structure is correct, and the base of the reef ball lies on the substrate with the largest opening at the top of the structure.
- 3) All reef material shall have all steel reinforcement rods, rebar, and other protrusions cut off and level with the surface of the concrete to minimize the snagging of fishing gear.
- 4) All reef materials must be clean and free from asphalt, petroleum, other hydrocarbons and toxic residues, plastics, Styrofoam, and other loose free-floating material, or other deleterious substances.
- 5) Materials shall be of sufficient weight in-water to not move from the site post-deployment.

#### Deployment:



- 6) Deployment shall be accomplished during daylight hours only.
- 7) Deployment activities will not commence until the project supervisor reports that no protected species have been sighted within 300 ft of the active deployment site (i.e., barge carrying material or moored vessel to be scuttled) for at least 20 minutes. Deployment activities will cease immediately if protected species are sighted within 300 ft of the active deployment site. Deployment activities will not recommence until the project supervisor reports that no protected species have been sighted for at least 20 minutes.

Vessel Movement:

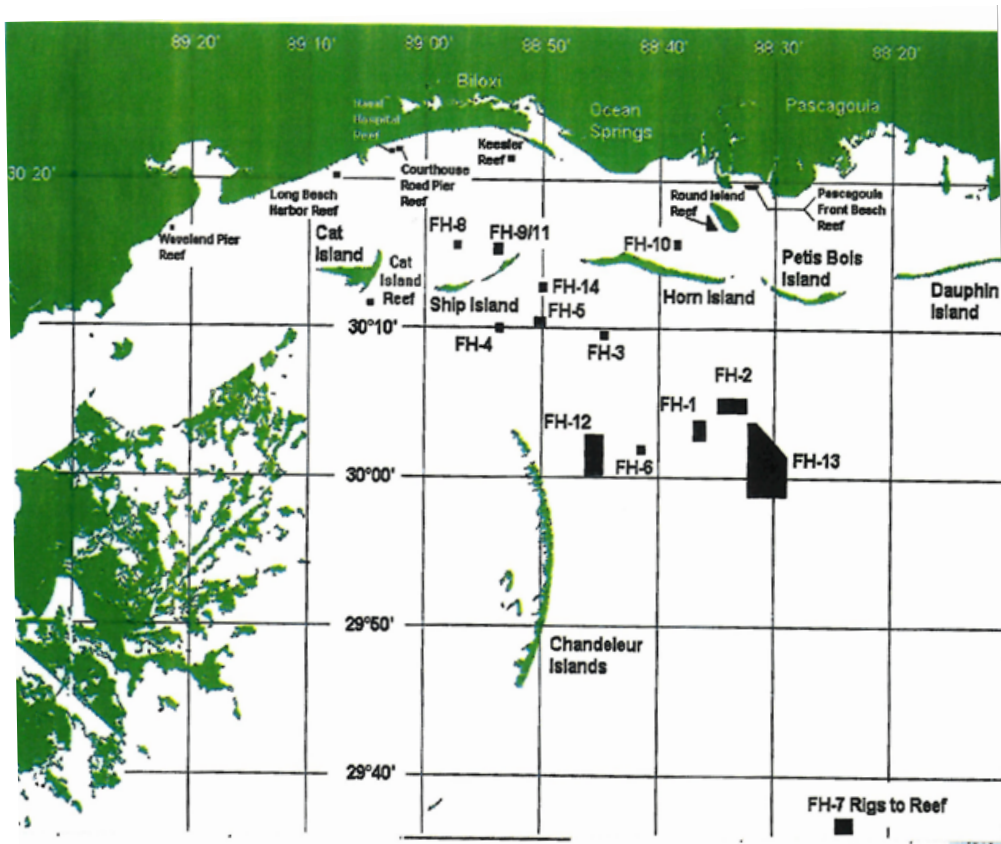
- 8) Personnel will be instructed of the potential presence of protected species and the need to avoid collision with them.
- 9) Personnel will be instructed that if a protected species is seen within 300 ft of vessel movement, all appropriate precautions shall be implemented to ensure its protection. These precautions shall include cessation of operation of any vessel closer than 50 ft of a protected species. Activities may not resume until the protected species has departed the project area of its own volition.
- 10) Personnel will be notified that all vessels associated with the construction project shall operate at "no wake/idle" speeds at all times while in the construction area and while in water depths where the draft of the vessel provides less than a four-foot clearance from the bottom.
- 11) All vessels will follow deep-water routes (e.g., marked channels) whenever possible.
- 12) Deployment vessels (loaded) shall draw no more than one (1) ft less the depth of water in which the reef material will be deployed.

## 2.2 Action Area

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02). For the purposes of this consultation, the action area includes the 7 existing reef sites described in the proposed action, which are located within the Mississippi Sound (North of the barrier islands) and the Gulf of Mexico (South of the barrier islands), and the vessel transit routes to and from the sites (Table 1, Figure 1). The sites are composed of a muddy sand bottom, and are void of any coral or seagrasses. Water depths within the deployment area range from 11 to 140 ft, and minimum clearances from the top of the structure to the water surface range from 7 to 128 ft. The existing sites also include previously-permitted artificial low-relief and high-relief materials such as concrete, rubble, limestone pyramids, materials of design (reef balls), barges, and vessels. These reef sites are a part of the MDMR Artificial Reef Program. FH sites 8-11 are located in GSCH. Currently, 48% of FH-8, 45% of FH-9/11, and 30% of FH-10 are not developed as reefs and contain open muddy sandy bottom, which equates to a total of approximately 179 acres (ac) of open substrate.

**Table 1. Project Site Coordinates and Area of FH Artificial Reef Sites Evaluated Under the Proposed Action**

Site	Latitude/Longitude <sup>2</sup> (North American Datum 1983)	Acres	Material to be Deployed
FH-1	30.0683, -88.6050	564.92	Up to 5 vessels
FH-2	30.0933, -88.5467	1,516.83	Up to 6 vessels
FH-7	29.6250, -88.3917	707.39	Up to 4 vessels
FH-8	30.2683, -88.9600	29.29	Low-relief
FH-9/11	30.2667, -88.8933	358.77	Low-relief
FH-10	30.2667, -88.6483	10.25	Low-relief



**Figure 1. Image of the project locations and surrounding area (Image provided by USACE)**

<sup>2</sup> Latitudes and longitudes represent the Northeast corner of each site.

### 3. STATUS OF LISTED SPECIES AND CRITICAL HABITAT

Table 2 provides the effect determinations for ESA-listed species the USACE and/or NMFS believe may be affected by the proposed action.

**Table 2. Effects Determinations for Species (DPSs) the Action Agency and/or NMFS Believes May Be Affected by the Proposed Action**

Species	ESA Listing Status <sup>3</sup>	Action Agency Effect Determination	NMFS Effect Determination
<b>Sea Turtles</b>			
Green (North Atlantic [NA] distinct population segment [DPS])	T	NLAA	LAA
Green (South Atlantic [SA] DPS)	T	NLAA	LAA
Kemp's ridley	E	NLAA	LAA
Leatherback	E	NE	LAA
Loggerhead (Northwest Atlantic [NWA] DPS)	T	NLAA	LAA
Hawksbill	E	NLAA	NLAA
<b>Fish</b>			
Gulf sturgeon (Atlantic sturgeon, Gulf subspecies)	T	NLAA	NLAA
Oceanic whitetip shark	T	NE	NE
Giant manta ray	T	NLAA	NLAA
<b>Marine Mammals</b>			
Gulf of Mexico Bryde's whale	E	NLAA	NE

We believe the project will have no effect on oceanic whitetip shark and Bryde's whale. Oceanic whitetip sharks are pelagic and found in deep, offshore oceanic waters. Additionally, Bryde's whales are generally found in the Bryde's whale core distribution area along the 100-m depth continental shelf contour and deeper water including canyons along the shelf break. As described above, the action area of the proposed action is within the Mississippi Sound (North of the barrier islands) and the Gulf of Mexico (South of the barrier islands), and the water depths within the deployment area range from 11 to 140 ft, which is not believed to be used by oceanic whitetip sharks or Bryde's whales.

<sup>3</sup> E = endangered; T = threatened; LAA = likely to adversely affect; NE = no effect; NLAA = may affect, not likely to adversely affect

**Table 3. Effects Determinations for Designated Critical Habitat the Action Agency and/or NMFS Believe Bay Be Affected by the Proposed Action**

Species	Critical Habitat Unit	USACE Effect Determination	NMFS Effect Determination
Gulf sturgeon	Lake Pontchartrain – Mississippi Sound – Unit 8	NLAA	LAA

### 3.1 Potential Routes of Effect Not Likely To Adversely Affect Listed Species

We have determined that the following potential routes of effect under the proposed action being considered in this Opinion are not likely to adversely affect (NLAA) hawksbill sea turtles, Gulf sturgeon, and giant manta ray. The potential routes of effects to these species are discussed below. Unlike the other ESA-listed sea turtles, hawksbill sea turtles are not likely to be adversely affected by the proposed action. Sea turtle stranding data indicate that hawksbill sea turtles have been found to occasionally wander in this area of the Gulf, but are not resident to this area. Specifically, 2007-2015 inshore and offshore stranding network data for the state of Alabama (zones 10 and 11) shows only 1 hawksbill sea turtle stranding. As a result, hawksbills would be so rare at the project sites that it is extremely unlikely they would be found interacting with artificial reef material. Therefore NMFS believes that the project activities may affect, but are not likely to adversely affect the hawksbill sea turtle as opposed to the other ESA-listed sea turtles. References to ESA-listed sea turtles below excludes hawksbills.

#### 3.1.1 *Effects of Habitat Exclusion*

ESA-listed species, namely sea turtles, Gulf sturgeon, and giant manta ray, might be adversely affected by their inability to access the project sites for foraging, refuge, and/or nursery habitat due to their avoidance of construction activities and related noise. We have determined these effects to be insignificant. Species may forage in the area but the size of the area from which animals will be excluded is relatively small in comparison to the available sandy habitat nearby. In addition, any disturbances to ESA-listed species would be intermittent (1 to 2 days per deployment opportunity) and construction will be limited to daylight hours only, and therefore the species will be able to move around the project site at night.

#### 3.1.2 *Effects of Material Deployment*

ESA-listed species, namely sea turtles, Gulf sturgeon, and giant manta ray, could be physically injured if struck by transport vessels or material during deployment at reef sites. We believe this is extremely unlikely to occur for the following reasons. All of these animals are highly mobile, and able to avoid slow-moving equipment. Furthermore, the PDCs require that deployment activities will cease immediately if any protected species is sighted within 300 ft (100 yds) of the active deployment site, and such activities will not recommence until the project supervisor reports that no protected species have been sighted for at least 20 minutes. If a protected species is seen within 300 ft (100 yds) of a project vessel, all appropriate precautions shall be implemented to avoid a collision. These precautions shall include ceasing any vessel movement when closer than 50 ft of a protected species (excluding at times when movement is required for

safe navigation [e.g., transiting inlets]). Operation will not resume until the protected species has departed the project area of its own volition.

### *3.1.3 Effects Related to the Permanent Loss of Habitat*

The installation of these artificial reefs will result in the permanent loss of unconsolidated bottom habitat that could be used by ESA-listed species for foraging. We believe the effect of the permanent habitat loss will be insignificant for sea turtles, Gulf sturgeon, and giant manta rays, given the mobility of these ESA-listed species and the remaining undeveloped habitat around the artificial reefs to utilize for feeding. Furthermore, this unconsolidated bottom habitat are not preferred foraging areas for these species. For example, benthic feeding sea turtles prey on encrusting organisms such as sponges, tunicates, corals, gorgonians, and algae not commonly found on unconsolidated bottom. Giant manta rays are filter feeders and primarily feed on surface zooplankton (Burgess et al. 2016; Couturier et al. 2013).

Of the sites covered under this Opinion, sites FH-8, FH-9, FH-10 and FH-11 may have adverse effects to GSCH, as discussed in Section 5.2.

### *3.1.4 Effects Related to Entrapment in and Entanglement with Low-Relief and High-Relief Artificial Reef Material and Fishing Gear*

ESA-listed species may be physically injured or killed if they become entangled in abandoned fishing gear or other debris that may accumulate on low-relief and high-relief artificial reefs, and ESA-listed sea turtles may become entrapped (stuck) in an artificial reef structure. For the reasons discussed below, we believe all ESA-listed species considered in this Opinion are extremely unlikely to become entangled or entrapped in low relief artificial reef material and that ESA-listed fish species are extremely unlikely to become entangled or entrapped in high relief artificial reef materials. As discussed further in Section 5, we believe entanglement in materials associated with high-relief artificial reefs may adversely affect sea turtles.

Low-relief and/or solid concrete material, rock rubble, and individual artificial reef modules present less complicated vertical relief that is not as likely to accumulate monofilament as larger, higher-relief materials, as documented in Barnette (2017). The implementation of the PDCs 2, 3 and 7 listed above in Section 2.1 would further reduce the likelihood of entanglement and entrapment. PDC 2 requires that materials of design, such as "reef balls" used for offshore deployments, are to have a solid base of approximately 6 ft in diameter and a height of 5 ft. The reef ball will have a minimum of 18 openings along its surface. The top of the reef ball will be required to have a minimum of a 42-inch (in) opening. The reef balls will be deployed in a manner that ensures the placement of structure is correct, and the base of the reef ball lies on the substrate with the largest opening at the top of the structure. PDC 3 requires that all reef material have all steel reinforcement rods, rebar, and other protrusions cut off and level with the surface of the concrete to minimize the snagging of fishing gear. Furthermore, as described above, PDC 7 requires that deployment activities will not commence until the project supervisor reports that no protected species have been sighted within 300 ft of the active deployment site for at least 20 minutes and to cease all deployment activities immediately if any protected species are sighted within 300 ft of the active deployment site. Deployment activities will not recommence until the project supervisor reports that no protected species have been sighted for at least 20 minutes. The

best available information presented in Barnette (2017) indicates that gear and animal entanglement and sea turtle entrapment on low-relief material is extremely unlikely to occur under these conditions.

With respect to high-relief artificial reef material, we do not anticipate ESA-listed species to experience entrapment. We anticipate that only sea turtles are likely to experience entanglement events. Entanglement of the ESA-listed fish species that may be in the action area is extremely unlikely to occur. We have no information documenting any artificial reef entanglement event involving these fish species and it is also extremely unlikely that these species will utilize artificial reefs as habitat, as these species do not typically feed or rest on/near artificial reef structures due to their life history patterns, thus decreasing any potential for interactions with accumulated monofilament. Life history patterns also make it unlikely for sea turtles to become entrapped in high relief structures. On the other hand, high-relief artificial reef material has been shown to have adverse effects on sea turtles due to potential entanglement. Those effects from the proposed actions on Kemp's ridley, loggerhead, green, and leatherback sea turtles are discussed further in Section 5.

ESA-listed species could also be injured or killed as a result of hooking or other interactions incidental to fishing activities in the vicinity of the proposed action; however, the proposed action is extremely unlikely to increase the risk of incidental capture. There is no evidence that establishment of artificial reefs increases the numbers of fishers or boats participating in a given fishery.

### **3.2 Status of Species Likely To Be Adversely Affected**

The following subsections are synopses of the best available information on the statuses of the sea turtle species that are likely to be adversely affected by one or more components of the proposed action, including species-specific information on the distribution, population structure, life history, abundance, and population trends, and threats. The biology and ecology of these species as well as their status and trends inform the effects analysis for this Opinion.

#### **3.2.1 Sea Turtles**

Green [NA and SA DPSs], Kemp's ridley, leatherback, and loggerhead [NWA DPS] sea turtles travel widely throughout the South Atlantic, Gulf of Mexico and the Caribbean, and may be adversely affected by the proposed action. Section 3.2.1.1 of this Opinion will address the general threats that confront all sea turtle species. The remainder of Section 3.2.1 (Sections 3.2.1.2 – 3.2.1.5) will address information on the distribution, life history, population structure, abundance, population trends, and unique threats to the species of sea turtles further discussed in this Opinion.

##### **3.2.1.1 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, those identified in this section are discussed in a general sense for all sea turtles.

Threat information specific to a particular species are then discussed in the corresponding status sections where appropriate.

### Fisheries

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

In addition to domestic fisheries, sea turtles are subject to direct as well as incidental capture in numerous foreign fisheries, further impeding the ability of sea turtles to survive and recover on a global scale. For example, pelagic stage sea turtles, especially loggerheads and leatherbacks, circumnavigating the Atlantic are susceptible to international longline fisheries including the Azorean, Spanish, and various other fleets (Aguilar et al. 1995; Bolten et al. 1994). Bottom longlines and gillnet fishing is known to occur in many foreign waters, including (but not limited to) the northwest Atlantic, western Mediterranean, South America, West Africa, Central America, and the Caribbean. Shrimp trawl fisheries are also occurring off the shores of numerous foreign countries and pose a significant threat to sea turtles similar to the impacts seen in U.S. waters. Many unreported takes or incomplete records by foreign fleets make it difficult to characterize the total impact that international fishing pressure is having on listed sea turtles. Nevertheless, international fisheries represent a continuing threat to sea turtle survival and recovery throughout their respective ranges.

### Non-Fishery In-Water Activities

There are also many non-fishery impacts affecting the status of sea turtle species, both in the ocean and on land. In nearshore waters of the United States, the construction and maintenance of federal navigation channels has been identified as a source of sea turtle mortality. Hopper dredges, which are frequently used in ocean bar channels and sometimes in harbor channels and offshore borrow areas, move relatively rapidly and can entrain and kill sea turtles (NMFS 1997). Sea turtles entering coastal or inshore areas have also been affected by entrainment in the cooling-water systems of electrical generating plants. Other nearshore threats include harassment and/or injury resulting from private and commercial vessel operations, military detonations and training exercises, in-water construction activities, and scientific research activities.

### Coastal Development and Erosion Control

Coastal development can deter or interfere with nesting, affect nesting success, and degrade nesting habitats for sea turtles. Structural impacts to nesting habitat include the construction of

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

The April 20, 2010, explosion of the Deepwater Horizon (DWH) oil rig affected sea turtles in the Gulf of Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. The spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. Information on the spill impacts to individual sea turtle species is presented in the Status of the Species sections for each species.

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

### Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change, exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and



change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see <http://www.climate.gov>).

Climate change impacts on sea turtles currently cannot be predicted with any degree of certainty; however, significant impacts to the hatchling sex ratios of sea turtles may result (NMFS and USFWS 2007b). 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 degrees Celsius (°C) (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007b).

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 (National Research Council 1990b). These impacts will be exacerbated by sea level rise. If females nest on the seaward side of the erosion control structures, nests may be exposed to repeated tidal overwash (NMFS and USFWS 2007b). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005). The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

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

#### Other Threats

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

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

#### **3.2.1.2 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 and Wright (compilers) 1982; Turtle Expert Working Group 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 lbs (45 kilograms [kg]) and have a carapace length of around 2.1 ft (65 centimeters [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 millimeters [mm]) straight carapace length (SCL), 1.26-1.73 in (32-44 mm) in width, and 0.3-0.4 lb (15-20 grams [g]) in weight. Their return to nearshore coastal habitats typically occurs around 2 years of age (Ogren 1989a), although the time spent in the oceanic zone may vary from 1-4 years or perhaps more (TEWG 2000). Juvenile Kemp's ridley sea turtles use these nearshore coastal habitats from April through November, but they move towards more suitable overwintering habitat in deeper offshore waters (or more southern waters along the Atlantic coast) as water temperature drops.

The average rates of growth may vary by location, but generally fall within  $2.2-2.9 \pm 2.4$  in per year ( $5.5-7.5 \pm 6.2$  cm/year) (Schmid and Barichivich 2006; Schmid and Woodhead 2000). Age

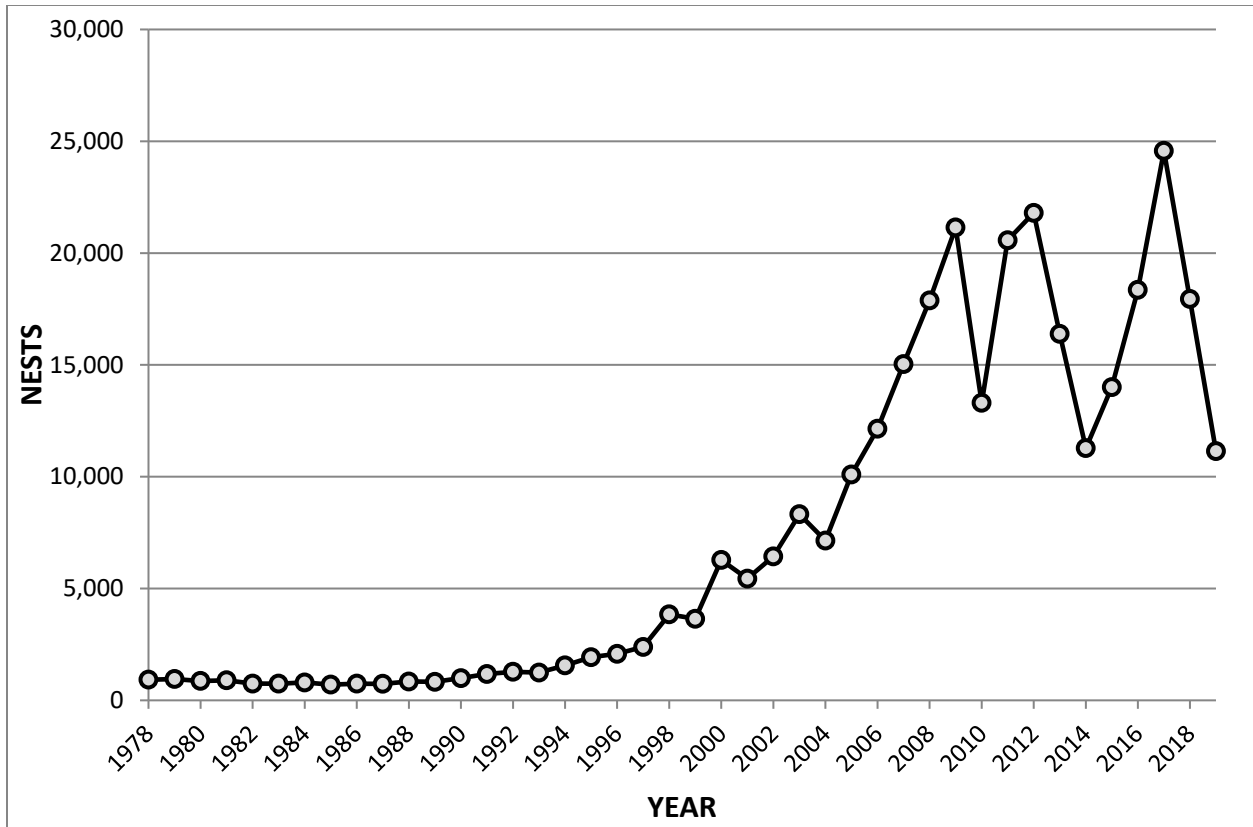
to sexual maturity ranges greatly from 5-16 years, though NMFS et al. (2011) determined the best estimate of age to maturity for Kemp's ridley sea turtles was 12 years. It is unlikely that most adults grow very much after maturity. While some sea turtles nest annually, the weighted mean remigration rate 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 2), 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 (Burchfield 2013). From 2013 through 2014, there was a second significant decline, as only 16,385 and 11,279 nests were recorded, respectively. More recent data, however, indicated an increase in nesting. In 2015 there were 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Gladys Porter Zoo 2016). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm., August 31, 2017), but nesting for 2018 has declined to 17,945, followed by another steep drop to 11,090 in 2019 (Gladys Porter Zoo database 2019). At this time, it is unclear whether the increases and declines in nesting seen over the past decade represents a population oscillating around an equilibrium point or if nesting will decline or increase in the future.

A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (National Park Service [NPS] data). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015, the record nesting in 2017, and then a drop back down to 190 nests in 2019 (NPS data).



**Figure 2.** 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 (Turtle Expert Working Group 1998a; Turtle Expert Working Group 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 and 2018-2019 potentially indicate a serious population-level impact, and there is cause for concern regarding the ongoing recovery trajectory.

## Threats

Kemp's ridley sea turtles face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1.1; 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*<sup>4</sup> are increasingly established, bacterial and fungal pathogens in nests are also likely to increase. Bacterial and fungal pathogen impacts have been well documented in the large arribadas of the olive ridley at Nancite in Costa Rica (Mo 1988). In some years, and on some sections of the beach, the hatching success can be as low as 5% (Mo 1988). As the Kemp's ridley nest density at Rancho Nuevo and adjacent beaches continues to increase, appropriate monitoring of emergence success will be necessary to determine if there are any density-dependent effects.

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

Nonetheless, considering that strandings typically represent only a small fraction of actual mortality, these stranding events potentially represent a serious impact to the recovery and survival of the local sea turtle populations. While a definitive cause for these strandings has not been identified, necropsy results indicate a significant number of stranded turtles from these events likely perished due to forced submergence, which is commonly associated with fishery

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<sup>4</sup> *Arribada* is the Spanish word for "arrival" and is the term used for massive synchronized nesting within the genus *Lepidochelys*.

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 turtle excluder devices (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 Federal Register [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. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of Mexico may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 3.2.1.1, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles are considered here. Kemp's ridleys experienced the greatest negative impact stemming from the DWH oil spill event of any sea turtle species. Impacts to Kemp's ridley sea turtles occurred to offshore small juveniles, as well as large juveniles and adults. Loss of hatchling production resulting from injury to adult turtles was also estimated for this species. Injuries to adult turtles of other species, such as loggerheads, certainly would have resulted in unrealized nests and hatchlings to those species as well. Yet, the calculation of unrealized nests and hatchlings was limited to Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of Mexico throughout their lives (DWH Trustees 2016).

A total of 217,000 small juvenile Kemp's ridleys (51.5% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. That means approximately half of all small juvenile Kemp's ridleys from the total population estimate of 430,000 oceanic small juveniles were exposed to oil. Furthermore, a large number of small juveniles were removed from the population, as up to 90,300 small juveniles Kemp's ridleys are estimated to have died as a direct result of the exposure. Therefore, as much as 20% of the small

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

### **3.2.1.3 Loggerhead Sea Turtle (NWA DPS)**

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

#### *Species Description and Distribution*

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

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

The majority of loggerhead nesting occurs at the western rims of the Atlantic and Indian Oceans concentrated in the north and south temperate zones and subtropics (National Research Council 1990a). For the NWA DPS, most nesting occurs along the coast of the United States, from southern Virginia to Alabama. Additional nesting beaches for this DPS are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba (Moncada-Gavilán 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 (Turtle Expert Working Group 1998a).

Within the NWA DPS, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf Coast of Florida. Previous Section 7 analyses have recognized at least 5 western Atlantic subpopulations, divided geographically as follows: (1) a Northern nesting subpopulation, occurring from North Carolina to northeast Florida at about 29°N; (2) a South Florida nesting subpopulation, occurring from 29°N on the east coast of the state to Sarasota on the west coast; (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez M. 1990; Turtle Expert Working Group 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 [NRU] (Florida/Georgia border north through southern Virginia), (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The recovery plan concluded that all recovery units are essential to the recovery of the species. Although the recovery plan was written prior to the listing of the NWA DPS, the recovery units for what was then termed the Northwest Atlantic population apply to the NWA DPS.

### *Life History Information*

The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, which include the ecosystems those stages generally use: (1) egg



(terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone<sup>5</sup>), (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 ounces (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 the Indian River Lagoon, Florida, are regularly used by juveniles but not by adult loggerheads. Adult loggerheads do tend to use estuarine areas with more open ocean access, such as the Chesapeake Bay in the U.S. mid-Atlantic. Shallow-water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads (Conant et al. 2009).

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<sup>5</sup> Neritic refers to the nearshore marine environment from the surface to the sea floor where water depths do not exceed 200 meters.

Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has also been documented (Hawkes et al. 2007) Georgia Department of Natural Resources [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. 2003a; NMFS 2001; NMFS 2009; NMFS and USFWS 2008; Turtle Expert Working Group 1998a; Turtle Expert Working Group 2000; Turtle Expert Working Group 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

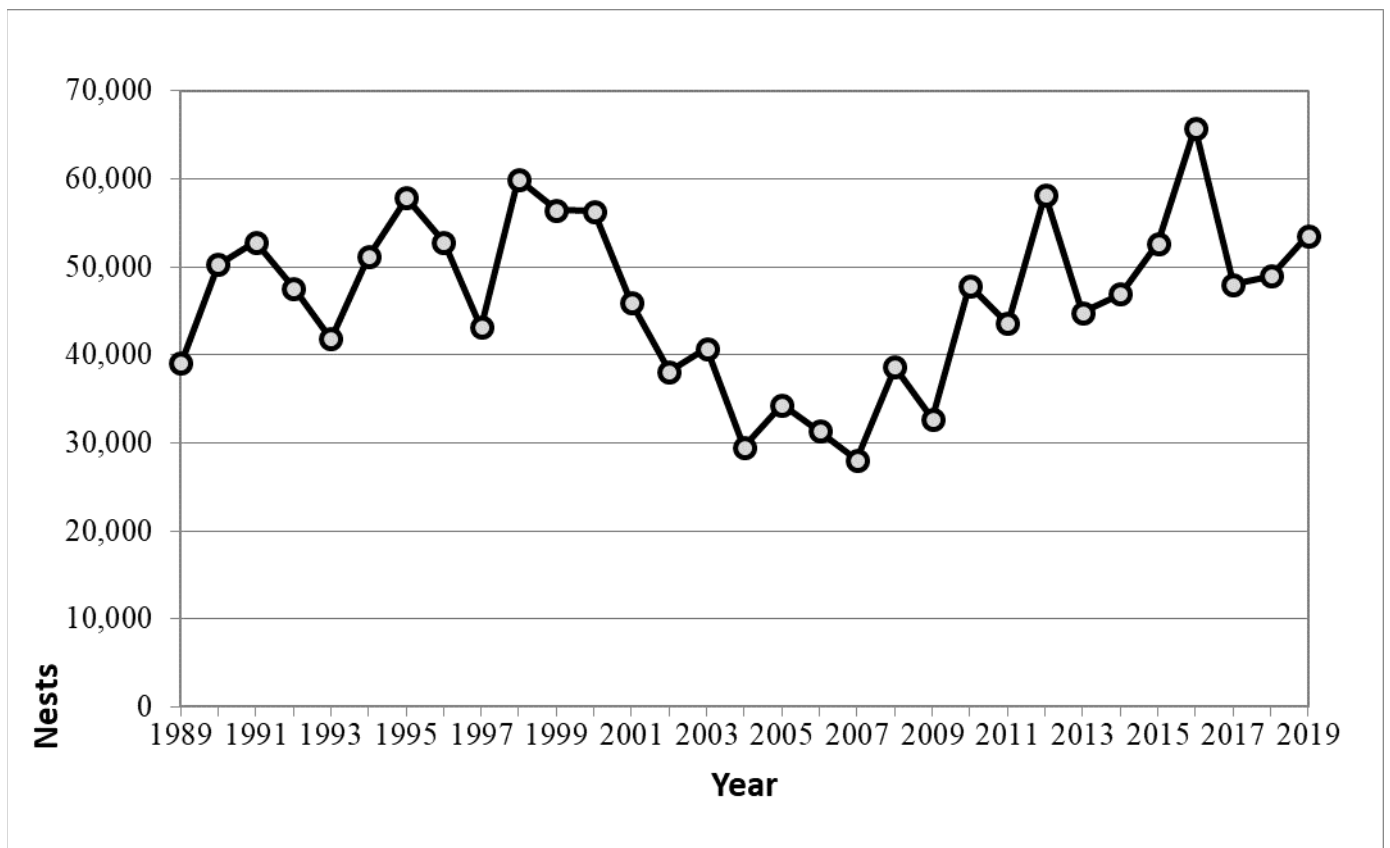
Numbers of nests and nesting females can vary widely from year to year. Nesting beach surveys, though, can provide a reliable assessment of trends in the adult female population, due to the strong nest site fidelity of female loggerhead sea turtles, as long as such studies are sufficiently long and survey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

#### Peninsular Florida Recovery Unit

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

In addition to the total nest count estimates, the FWRI uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. This provides a better tool for understanding the nesting trends (Figure 3). The FWRI has annually collected loggerhead index nesting beach survey totals in Florida beginning in 1989 (Ceriani et al. 2019). Over that time period, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that

was followed by a sharp decline that lasted from 1999-2007. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represented a new record for loggerheads on the core index beaches. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but nonsignificant increasing trend. The data from 1989 through 2016 exhibits an overall positive change in the nest counts although there is a wide variability between 2012-2016 resulting in widening confidence intervals (Ceriani, et al, 2019). Nesting at the core index beaches declined in 2017 to 48,033, and rose slightly again to 48,983 in 2018 and then to 53,507 in 2019, which is the 3<sup>rd</sup> highest total since 2001. However, given the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.) it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani, et al. 2019).



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

### Northern Recovery Unit

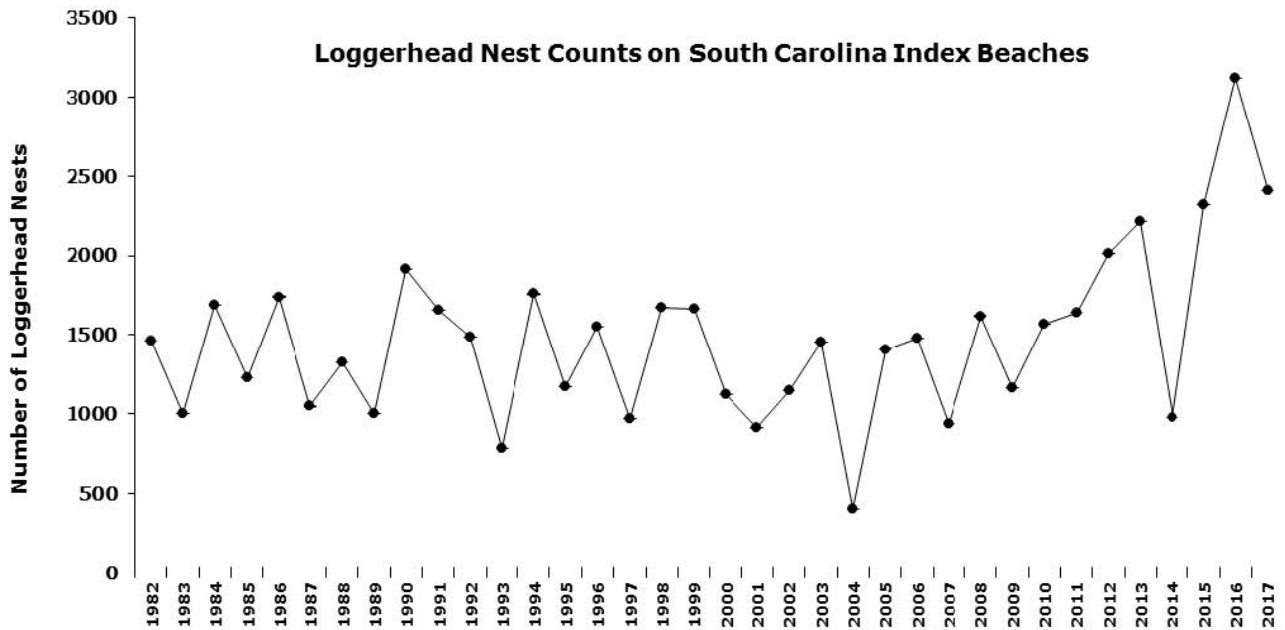
Annual nest totals from beaches within the NRU averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches (GADNR unpublished data, North Carolina Wildlife Resources Commission [NCWRC] unpublished data, SCDNR unpublished data), and represent approximately 1,272 nesting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

Data since that analysis (Table 4) are showing improved nesting numbers in Georgia, South Carolina, and North Carolina. Loggerhead nesting in Georgia, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016. Nesting in 2017 and 2018 declined relative to 2016, back to levels seen in 2013 to 2015, but then bounced back in 2019, breaking records for each of the three states and the overall Recovery Unit.

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

South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record (Figure 4). South Carolina has not updated their Index Beach information, but it likely follows a similar pattern to the statewide data in Table 4 above.



**Figure 4. South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: <http://www.dnr.sc.gov/seaturtle/nest.htm>)**

Other NWA DPS Recovery Units

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

In-water Trends

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

(Arendt et al. 2009; Ehrhart et al. 2007; Epperly et al. 2007). Researchers believe that this increase in CPUE is likely linked to an increase in juvenile abundance, although it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. Bjorndal et al. (2005), cited in NMFS and USFWS (2008), caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern United States may be due to increased abundance of the largest oceanic/neritic juveniles (historically referred to as small benthic juveniles), which could indicate a relatively large number of individuals around the same age may mature in the near future (Turtle Expert Working Group 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 (Turtle Expert Working Group 2009).

#### Population Estimate

The NMFS Southeast Fisheries Science Center developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS 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). Another estimate for the entire western North Atlantic population was a mean of 38,334 adult females using data from 2001-2010 (Richards et al. 2011). A 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 (Specific to Loggerhead Sea Turtles)*

The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 3.2.1.1. Yet the impact of fishery interactions is a point of further emphasis for this species. The joint NMFS and USFWS Loggerhead Biological Review Team determined that the greatest threats to the NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant et al. 2009).

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

Storelli et al. (2008) analyzed tissues from stranded loggerhead sea turtles and found that mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991).

While oil spill impacts are discussed generally for all species in Section 3.2.1.1, 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 (Deepwater Horizon Natural Resource Damage Assessment 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.

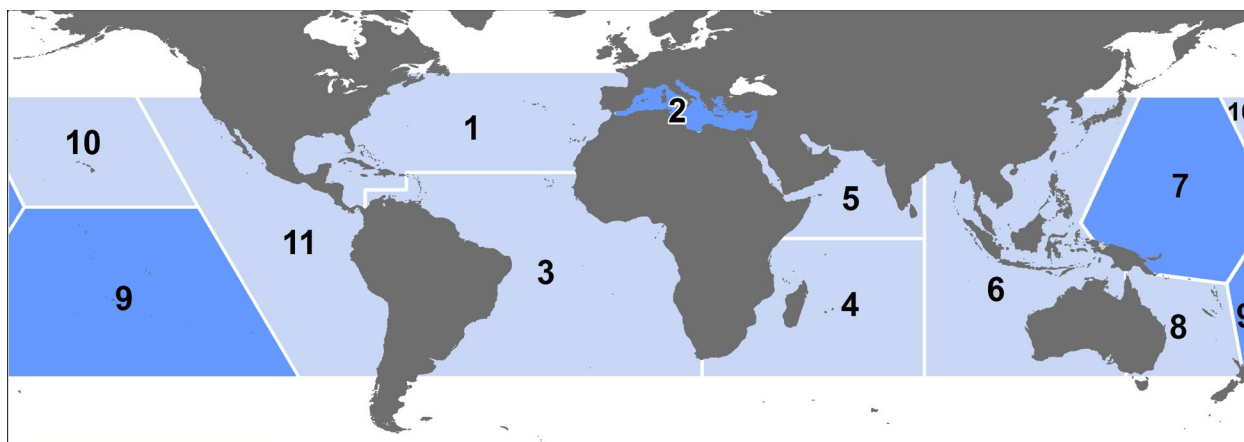
Unlike Kemp's ridleys, the majority of nesting for the NWA DPS occurs on the Atlantic coast, and thus loggerheads were impacted to a relatively lesser degree. However, it is likely that impacts to the NGMRU of the NWA DPS would be proportionally much greater than the impacts occurring to other recovery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

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

(Hawkes et al. 2007; Weishampel et al. 2004), short inter-nesting intervals (Hays et al. 2002), and shorter nesting seasons (Pike et al. 2006).

### 3.2.1.4 Green Sea Turtle (Information Relevant to All DPSs)

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 distinct population segments (DPSs) (81 FR 20057 2016) (Figure 5). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific DPSs were listed as threatened. For the purposes of this consultation, only the South Atlantic DPS (SA DPS) and North Atlantic DPS (NA DPS) will be considered, as they are the only two DPSs with individuals occurring in the Atlantic and Gulf of Mexico waters of the United States.



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

#### *Species Description and Distribution*

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

With the exception of post-hatchlings, green sea turtles live in nearshore tropical and subtropical waters where they generally feed on marine algae and seagrasses. They have specific foraging grounds and may make large migrations between these forage sites and natal beaches for nesting (Hays et al. 2001). Green sea turtles nest on sandy beaches of mainland shores, barrier islands, coral islands, and volcanic islands in more than 80 countries worldwide (Hirth 1997). The 2



largest nesting populations are found at Tortuguero, on the Caribbean coast of Costa Rica (part of the NA DPS), and Raine Island, on the Pacific coast of Australia along the Great Barrier Reef. Differences in mitochondrial DNA properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; FitzSimmons et al. 2003). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Within U.S. waters individuals from both the NA and SA DPSs can be found on foraging grounds. While there are currently no in-depth studies available to determine the percent of NA and SA DPS individuals in any given location, two small-scale studies provide an insight into the degree of mixing on the foraging grounds. An analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of Mexico) found approximately 4% of individuals came from nesting stocks in the SA DPS (specifically Suriname, Aves Island, Brazil, Ascension Island, and Guinea Bissau) (Foley et al. 2007). On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the Aves Island/Suriname nesting assemblage, which is part of the SA DPS (Bass and Witzell 2000). All of the individuals in both studies were benthic juveniles. Available information on green turtle migratory behavior indicates that long distance dispersal is only seen for juvenile turtles. This suggests that larger adult-sized turtles return to forage within the region of their natal rookeries, thereby limiting the potential for gene flow across larger scales (Monzón-Argüello et al. 2010). While all of the mainland U.S. nesting individuals are part of the NA DPS, the U.S. Caribbean nesting assemblages are split between the NA and SA DPS. Nesters in Puerto Rico are part of the NA DPS, while those in the U.S. Virgin Islands are part of the SA DPS. We do not currently have information on what percent of individuals on the U.S. Caribbean foraging grounds come from which DPS.

#### North Atlantic DPS Distribution

The NA DPS boundary is illustrated in Figure 5. 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 (NMFS and USFWS 1991; Piniak et al. 2007). 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 1991; 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 1997), and the northwestern coast of the Yucatán Peninsula.

### South Atlantic DPS Distribution

The SA DPS boundary is shown in Figure 5, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

The in-water range of the SA DPS is widespread. In the eastern South Atlantic, significant sea turtle habitats have been identified, including green turtle feeding grounds in Corisco Bay, Equatorial Guinea/Gabon (Formia 1999); Congo; Mussulo Bay, Angola (Carr and Carr 1991); as well as Principe Island. Juvenile and adult green turtles utilize foraging areas throughout the Caribbean areas of the South Atlantic, often resulting in interactions with fisheries occurring in those same waters (Piniak et al. 2007). Juvenile green turtles from multiple rookeries also frequently utilize the nearshore waters off Brazil as foraging grounds as evidenced from the frequent captures by fisheries (Lima et al. 2010; López-Barrera et al. 2012; Marcovaldi et al. 2009). Genetic analysis of green turtles on the foraging grounds off Ubatuba and Almofala, Brazil show mixed stocks coming primarily from Ascension, Suriname and Trindade as a secondary source, but also Aves, and even sometimes Costa Rica (North Atlantic DPS)(Naro-Maciel et al. 2007; Naro-Maciel et al. 2012). While no nesting occurs as far south as Uruguay and Argentina, both have important foraging grounds for South Atlantic green turtles (Gonzalez Carman et al. 2011; Lezama 2009; López-Mendilaharsu et al. 2006; Prosdocimi et al. 2012; 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 1979; 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 oz (25 g). 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) (Campbell and Lagueur 2005; Chaloupka and Limpus 2005).

After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. This early oceanic phase remains one of the most poorly understood aspects of green sea turtle life history (NMFS and USFWS 2007a). Green sea turtles exhibit particularly slow growth rates of about 0.4-2 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 (Ingle and Smith 1974), although some populations are known to also feed heavily on invertebrates (Carballo et al. 2002). Green sea turtles mature slowly, requiring 20-50 years to reach sexual maturity (Chaloupka and Musick 1997; Hirth 1997).

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

#### *Status and Population Dynamics*

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

#### North Atlantic DPS

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

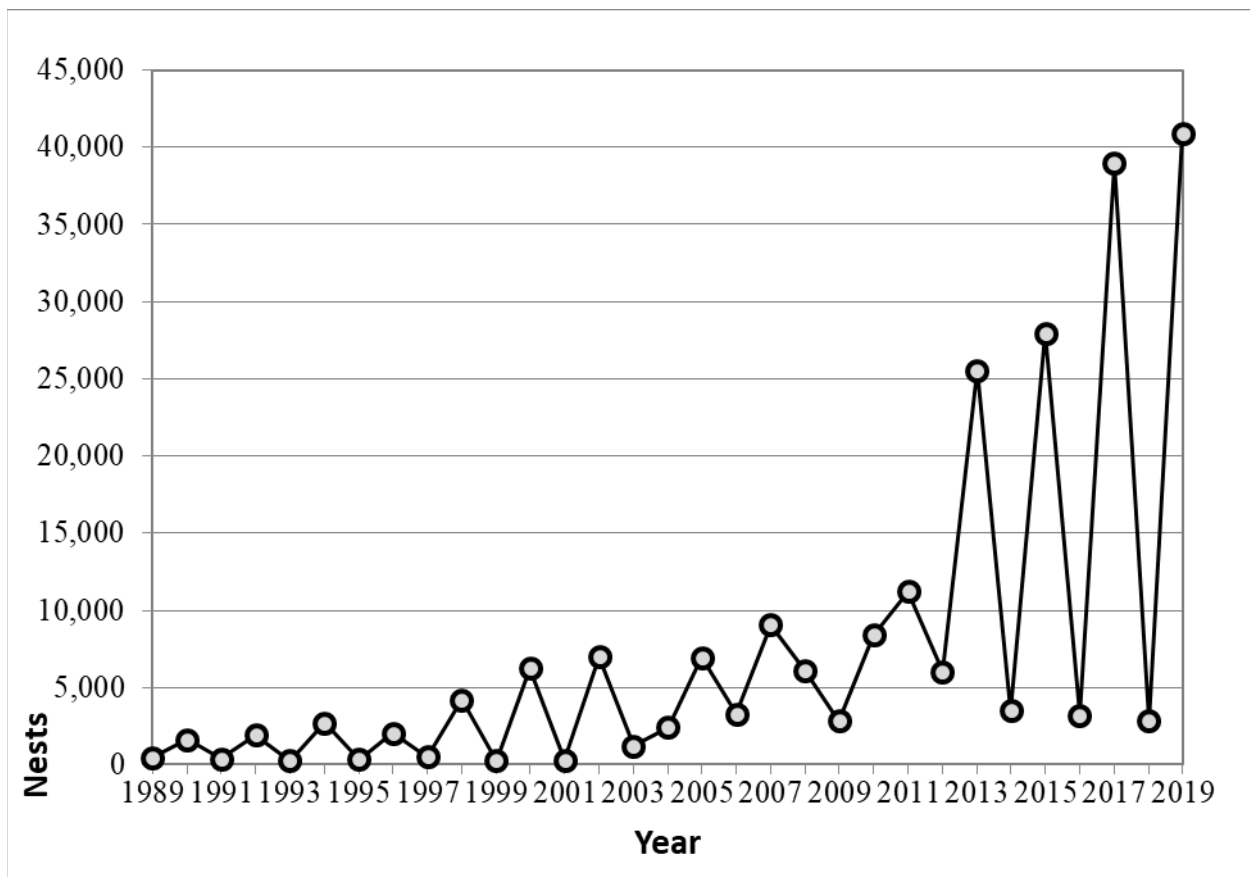
Quintana Roo, Mexico, accounts for approximately 11% of nesting for the DPS (Seminoff et al. 2015). In the early 1980s, approximately 875 nests/year were deposited, but by 2000 this

increased to over 1,500 nests/year (NMFS and USFWS 2007a). 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 1970s, when monitoring began. For instance, from 1971-1975 there were approximately 41,250 average annual emergences documented and this number increased to an average of 72,200 emergences from 1992-1996 (Bjorndal et al. 1999). Troëng and Rankin (2005) collected nest counts from 1999-2003 and also reported increasing trends in the population consistent with the earlier studies, with nest count data suggesting 17,402-37,290 nesting females per year (NMFS and USFWS 2007a). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually. Overall, 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 1998; Troëng and Rankin 2005).

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](http://www.seaturtle.org)).

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 6). According to data collected from Florida's index nesting beach survey from 1989-2018, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 40,911 in 2019. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011, and a return to the trend of biennial peaks in abundance thereafter (Figure 6). 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 (NWR) growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years.



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

Similar to the nesting trend found in Florida, in-water studies in Florida have also recorded increases in green turtle captures at the Indian River Lagoon site, with a 661 percent increase over 24 years (Ehrhart et al. 2007), and the St Lucie Power Plant site, with a significant increase in the annual rate of capture of immature green turtles (SCL<90 cm) from 1977 to 2002 or 26 years (3,557 green turtles total; M. Bressette, Inwater Research Group, unpubl. data; (Witherington et al. 2006). In Florida, most nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, FFWCC, pers. comm., 2013).

### South Atlantic DPS

The SA DPS is large, estimated at over 63,000 nesters, but data availability is poor. More than half of the 51 identified nesting sites (37) did not have sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). This includes some sites, such as beaches in French Guiana, which are suspected to have large numbers of nesters. For example, the nesting at Poilão, Guinea-Bissau, accounted for approximately 46% of that estimate (approximately 30,000 nesters), with Ascension Island, United Kingdom, (approximately 13,400 nesters; 21%), and the Galibi Reserve, Suriname (approximately 9,400 nesters; 15%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015). 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).

Seminoff et al. (2015) reported that while trends cannot be estimated for many nesting populations due to the lack of data, they could discuss possible trends at some of the primary nesting sites. Seminoff et al. (2015) indicated that the nesting concentration at Ascension Island (United Kingdom) is one of the largest in the SA DPS and the population has increased substantially over the last 3 decades (Broderick et al. 2006; Glen et al. 2006). Mortimer and Carr (1987) counted 5,257 nests in 1977 (about 1,500 females), and 10,764 nests in 1978 (about 3,000 females) whereas from 1999–2004, a total of about 3,500 females nested each year (Broderick et al. 2006). Since 1977, numbers of nests on 1 of the 2 major nesting beaches, Long Beach, have increased exponentially from around 1,000 to almost 10,000 (Seminoff et al. 2015). From 2010 to 2012, an average of 23,000 nests per year was laid on Ascension (Seminoff et al. 2015). Seminoff et al. (2015), caution that while these data are suggestive of an increase, historic data from additional years are needed to fully substantiate this possibility.

Seminoff et al. (2015) reported that the nesting concentration at Galibi Reserve and Matapica in Suriname was stable from the 1970s through the 1980s. From 1975–1979, 1,657 females were counted (Schulz 1982), a number that increased to a mean of 1,740 females from 1983–1987 (Ogren 1989b), and to 1,803 females in 1995 (Weijerman et al. 1996). Since 2000, there appears to be a rapid increase in nest numbers (Seminoff et al. 2015).

In the Bijagos Archipelago (Poilão, Guinea-Bissau), Parris and Agardy (1993 as cited in Fretey 2001) reported approximately 2,000 nesting females per season from 1990 to 1992, and Catry et al. (2002) reported approximately 2,500 females nesting during the 2000 season. Given the typical large annual variability in green sea turtle nesting, Catry et al. (2009) suggested it was premature to consider there to be a positive trend in Poilão nesting, though others have made such a conclusion (Broderick et al. 2006). Despite the seeming increase in nesting, interviews along the coastal areas of Guinea-Bissau generally resulted in the view that sea turtles overall have decreased noticeably in numbers over the past two decades (Catry et al. 2009). In 2011, a record estimated 50,000 green sea turtle clutches were laid throughout the Bijagos Archipelago (Seminoff et al. 2015).

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

### *Threats*

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green

sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1.1.

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 1995), and environmental conditions (e.g., habitat degradation, pollution, low wave energy, and shallow water (Foley et al. 2005). FP is cosmopolitan, but it has been found to affect large numbers of animals in specific areas, including Hawaii and Florida (Herbst 1994; Jacobson 1990; Jacobson et al. 1991).

Cold-stunning is another natural threat to green sea turtles. Although it is not considered a major source of mortality in most cases, as temperatures fall below 46.4°-50°F (8°-10°C) turtles may lose their ability to swim and dive, often floating to the surface. The rate of cooling that precipitates cold-stunning appears to be the primary threat, rather than the water temperature itself (Milton and Lutz 2003). Sea turtles that overwinter in inshore waters are most susceptible to cold-stunning because temperature changes are most rapid in shallow water (Witherington and Ehrhart 1989a). During January 2010, an unusually large cold-stunning event in the southeastern United States resulted in around 4,600 sea turtles, mostly greens, found cold-stunned, and hundreds found dead or dying. A large cold-stunning event occurred in the western Gulf of Mexico in February 2011, resulting in approximately 1,650 green sea turtles found cold-stunned in Texas. Of these, approximately 620 were found dead or died after stranding, while approximately 1,030 turtles were rehabilitated and released. During this same time frame, approximately 340 green sea turtles were found cold-stunned in Mexico, though approximately 300 of those were subsequently rehabilitated and released.

Whereas oil spill impacts are discussed generally for all species in Section 3.2.1.1, 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) (Deepwater Horizon Natural Resource Damage Assessment 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 (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016).

### **3.2.1.5 Leatherback Sea Turtle**

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

#### *Species Description and Distribution*

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

Unlike other sea turtles, leatherbacks have several unique traits that enable them to live in cold water. For example, leatherbacks have a countercurrent circulatory system (Greer Jr. et al. 1973),<sup>6</sup> a thick layer of insulating fat (Davenport et al. 1990; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),<sup>7</sup> and they can increase their body temperature through increased metabolic activity (Bostrom and Jones 2007; Southwood et al. 2005). These adaptations allow leatherbacks to be comfortable in a wide range of temperatures, which helps

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<sup>6</sup> Countercurrent circulation is a highly efficient means of minimizing heat loss through the skin's surface because heat is recycled. For example, a countercurrent circulation system often has an artery containing warm blood from the heart surrounded by a bundle of veins containing cool blood from the body's surface. As the warm blood flows away from the heart, it passes much of its heat to the colder blood returning to the heart via the veins. This conserves heat by recirculating it back to the body's core.

<sup>7</sup> "Gigantothermy" refers to a condition when an animal has relatively high volume compared to its surface area, and as a result, it loses less heat.



them to travel further than any other sea turtle species (NMFS and USFWS 1995). For example, a leatherback may swim more than 6,000 mi (10,000 km) in a single year (Benson et al. 2007a; Benson et al. 2011; Eckert 2006). They search for food between latitudes 71°N and 47°S in all oceans, and travel extensively to and from their tropical nesting beaches. In the Atlantic Ocean, leatherbacks have been recorded as far north as Newfoundland, Canada, and Norway, and as far south as Uruguay, Argentina, and South Africa (NMFS 2001).

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

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

#### *Life History Information*

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

The average size of reproductively active females in the Atlantic is generally 5-5.5 ft (150-162 cm) CCL (Benson et al. 2007a; Hirth et al. 1993; Starbird and Suarez 1994). Still, females as small as 3.5-4 ft (105-125 cm) CCL have been observed nesting at various sites (Stewart et al. 2007).

Female leatherbacks typically nest on sandy, tropical beaches at intervals of 2-4 years (García-Muñoz and Sarti 2000; McDonald and Dutton 1996; Spotila et al. 2000). Unlike other sea turtle

species, female leatherbacks do not always nest at the same beach year after year; some females may even nest at different beaches during the same year (Dutton et al. 2005; Eckert et al. 1989b; Keinath and Musick 1993; Steyermark et al. 1996). Individual female leatherbacks have been observed with fertility spans as long as 25 years (Hughes 1996). Females usually lay up to 10 nests during the 3-6 month nesting season (March through July in the United States), typically 8-12 days apart, with 100 eggs or more per nest (Eckert et al. 2012; Eckert et al. 1989b; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). Yet, up to approximately 30% of the eggs may be infertile (Eckert et al. 1989b; Eckert et al. 1984; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50% worldwide (Eckert et al. 2012), which is lower than the greater than 80% reported for other sea turtle species (Miller 1997). In the United States, the emergent success is higher at 54-72% (Eckert and Eckert 1990; Stewart and Johnson 2006; Tucker 1988). Thus the number of hatchlings in a given year may be less than the total number of eggs produced in a season. Eggs hatch after 60-65 days, and the hatchlings have white striping along the ridges of their backs and on the edges of the flippers. Leatherback hatchlings weigh approximately 1.5-2 oz (40-50 g), and have lengths of approximately 2-3 in (51-76 mm), with fore flippers as long as their bodies. Hatchlings grow rapidly, with reported growth rates for leatherbacks from 2.5-27.6 in (6-70 cm) in length, estimated at 12.6 in (32 cm) per year (Jones et al. 2011).

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

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

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

(Benson et al. 2007b; Bowlby et al. 1994; Graham 2009; Shenker 1984; Starbird et al. 1993; Suchman and Brodeur 2005).

### *Status and Population Dynamics*

The status of the Atlantic leatherback population had been less clear than the Pacific population, which has shown dramatic declines at many nesting sites (Martínez et al. 2007; Santidrián Tomillo et al. 2007; Spotila et al. 2000). This uncertainty resulted from inconsistent beach and aerial surveys, cycles of erosion, and reformation of nesting beaches in the Guianas (representing the largest nesting area). Leatherbacks also show a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species. Coordinated efforts of data collection and analyses by the leatherback TEWG helped to clarify the understanding of the Atlantic population status up through the early 2000's (Turtle Expert Working Group 2007). However, additional information for the Northwest Atlantic population has more recently shown declines in that population as well, contrary to what earlier information indicated (Northwest Atlantic Leatherback Working Group [NWALWG] 2018). A full status review covering leatherback status and trends for all populations worldwide is being finalized (2020).

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

Researchers believe the cyclical pattern of beach erosion and then reformation has affected leatherback nesting patterns in the Guianas. For example, between 1979 and 1986, the number of leatherback nests in French Guiana had increased by about 15% annually (NMFS 2001). This increase was then followed by a nesting decline of about 15% annually. This decline corresponded with the erosion of beaches in French Guiana and increased nesting in Suriname. This pattern suggests that the declines observed since 1987 might actually be a part of a nesting cycle that coincides with cyclic beach erosion in Guiana (Schulz 1975). Researchers think that the cycle of erosion and reformation of beaches may have changed where leatherbacks nest throughout this region. The idea of shifting nesting beach locations was supported by increased nesting in Suriname,<sup>8</sup> while the number of nests was declining at beaches in Guiana (Hilterman

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<sup>8</sup> Leatherback nesting in Suriname increased by more than 10,000 nests per year since 1999 with a peak of 30,000 nests in 2001.

et al. 2003). Though this information suggested the long-term trend for the overall Suriname and French Guiana population was increasing. A more recent cycle of nesting declines from 2008-2017, as high as 31% annual decline in the Awala-Yalimapo area of French Guiana and almost 20% annual declines in Guyana, has changed the long-term nesting trends in the region negative as described above (NWALWG 2018).

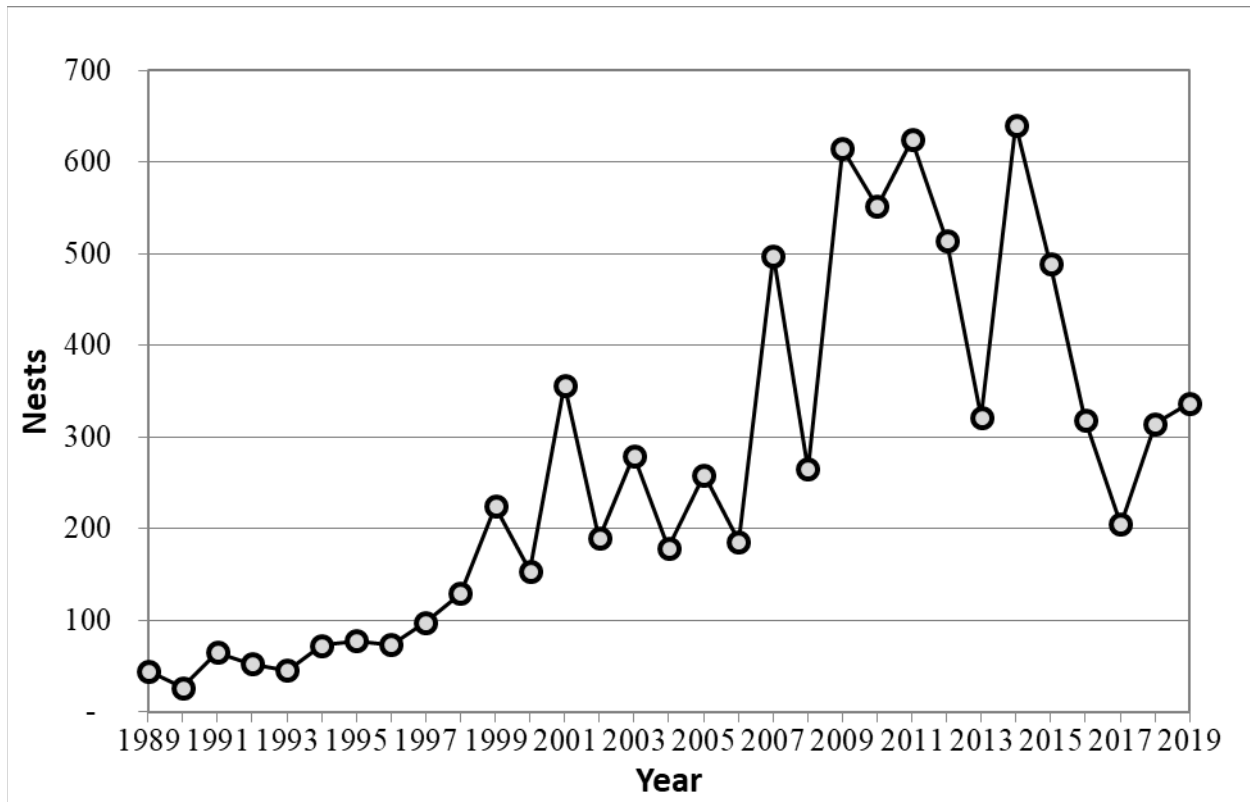
The Western Caribbean stock includes nesting beaches from Honduras to Colombia. Across the Western Caribbean, nesting is most prevalent in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coastline of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from index nesting beaches in Tortuguero, Gandoca, and Pacuaré in Costa Rica indicate that the nesting population likely was not growing over the 1995-2005 time series (Turtle Expert Working Group 2007). In Costa Rica, Tortuguero, leatherback nesting has decreased 88.5% overall from 1995 through 2011 (NMFS and USFWS 2013b). Other modeling of the nesting data for Tortuguero indicates a possible 67.8% decline between 1995 and 2006 (Troëng et al. 2007). Tiwari et al. (2013) report an estimated three-generation abundance change of -72%, -24%, and +6% for Tortuguero, Gandoca, and Pacuare, respectively. Further decline of almost 6% annual geometric mean from 2008-2017 reflects declines in nesting beaches throughout this stock (NWALWG 2018). Regardless of the method used to derive the estimate, the number of nests observed over the last 17 years has declined. Troëng et al. (2005) found a slight decline in the number of nests at Gandoca, Costa Rica, between 1995 and 2003, but the confidence intervals were large. Data between 1990 and 2004 at Gandoca averaged 582.9 (+ 303.3) nests each year, indicating nest numbers have been lower since 2000 (Chacón-Chaverri and Eckert 2007), and the numbers are not increasing (Turtle Expert Working Group 2007).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, St. Croix (U.S. Virgin Islands), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has increased from a minimum of 9 nests recorded in 1978 to between 469-882 nests recorded each year between 2000 and 2005, with an overall annual growth rate of 1.1% (Turtle Expert Working Group 2007). However since 2004, nesting has steadily declined in Culebra, which appears to reflect a shift in nest site fidelity rather than a decline in the female population (NMFS and USFWS 2013b). Tiwari et al. (2013) report an estimated three-generation abundance change of -4% and +5,583% at Culebra and Fajardo, respectively. At the primary nesting beach on St. Croix, the Sandy Point NWR, nesting has varied from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (Turtle Expert Working Group 2007). However, nesting data from 2001 through 2010 indicate nesting has slowed, possibly due to fewer new recruits and lowered reproductive output (NMFS and USFWS 2013b). From 2006-2010, Tiwari et al. (2013) report an annual growth rate of +7.5% in St. Croix and a three-generation abundance change of +1,058%. Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2% between 1994 and 2004 (Turtle Expert Working Group 2007). The nesting trend reversed course later, with an annual geometric mean decline of 10% from 2008-2017 driving the long-term trend (1990-2017) down to a 2% annual decline (NWALWG 2018).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission [FWC], unpublished data). Using data from the index nesting beach surveys, the Turtle Expert Working Group (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. The FWC Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 7 and Table 5). A similar pattern was also observed statewide (Table 5). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting. Overall, the trend showed growth on Florida’s east coast beaches. Tiwari et al. (2013) report an annual growth rate of 9.7% and a three-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, with a partial rebound in 2018 and 2019. The annual geometric mean trend for Florida has been a decline of almost 7% from 2008-2017, but the long-term trend (1990-2017) remains positive with an annual geometric mean increase of over 9% (NWALWG 2018).

**Table 5. Number of Leatherback Sea Turtle Nests in Florida**

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



**Figure 7. Leatherback sea turtle nesting at Florida index beaches since 1989**

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

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

Because the available nesting information is inconsistent, it is difficult to estimate the total population size for Atlantic leatherbacks. Spotila et al. (1996) characterized the entire Western Atlantic population as stable at best and estimated a population of 18,800 nesting females. Spotila et al. (1996) further estimated that the adult female leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, was about 27,600 (considering both nesting and interesting females), with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic determined by the Turtle Expert Working Group (2007). The Turtle Expert Working Group (2007) also determined that at the time of their publication, leatherback sea turtle populations in the Atlantic were all stable or increasing with the exception of the Western Caribbean and West Africa populations. A later review by NMFS and USFWS (2013a) suggested the leatherback nesting population was stable in most nesting regions of the Atlantic Ocean. However, as described earlier, the NW Atlantic population has experienced declines over the near term (2008-2017), often severe enough to reverse the longer term trends to negative where increases had previously been seen (NVALWG 2018). Given the relatively large size of the NW Atlantic population, it is likely that the overall Atlantic leatherback trend is no longer increasing.

### *Threats*

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

Of all sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, especially gillnet and pot/trap lines. This vulnerability may be because of their body type (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, their method of

locomotion, and/or their attraction to the lightsticks used to attract target species in longline fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine and many other stranded individuals exhibited evidence of prior entanglement (Dwyer et al. 2003). Zug and Parham (1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment from intense egg harvesting in some areas has caused a sharp decline in leatherback sea turtle populations. This represents a significant threat to survival and recovery of the species worldwide.

Leatherback sea turtles may also be more susceptible to marine debris ingestion than other sea turtle species due to their predominantly pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding and migratory purposes (Lutcavage et al. 1997; Shoop and Kenney 1992). The stomach contents of leatherback sea turtles revealed that a substantial percentage (33.8% or 138 of 408 cases examined) contained some form of plastic debris (Mrosovsky et al. 2009). Blocking of the gut by plastic to an extent that could have caused death was evident in 8.7% of all leatherbacks that ingested plastic (Mrosovsky et al. 2009). Mrosovsky et al. (2009) also note that in a number of cases, the ingestion of plastic may not cause death outright, but could cause the animal to absorb fewer nutrients from food, i.e., eat less in general; factors which could cause other adverse effects. The presence of plastic in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and forms of debris such as plastic bags (Mrosovsky et al. 2009). Balazs (1985) speculated that the plastic object might resemble a food item by its shape, color, size, or even movement as it drifts about, and therefore induce a feeding response in leatherbacks.

As discussed in Section 3.2.1.1, global climate change can be expected to have various impacts on all sea turtles, including leatherbacks. Global climate change is likely to also influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007c). Several studies have shown leatherback distribution is influenced by jellyfish abundance ((Houghton et al. 2006; Witt et al. 2007; Witt et al. 2006); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 3.2.1.1, specific impacts of the DWH oil spill on leatherback sea turtles are considered here. Available information indicates leatherback sea turtles (along with hawksbill turtles) were likely directly affected by the oil spill. Leatherbacks were documented in the spill area, but the number of affected leatherbacks was not estimated due to a lack of information compared to other species. But given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (TEWG 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, it was concluded that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. Potential DWH-related impacts to leatherback sea turtles include direct oiling or contact with dispersants from surface and subsurface oil and dispersants, inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

Although adverse impacts likely occurred to leatherbacks, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event may be relatively low. Thus, a population-level impact may not have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

### **3.3 Critical Habitat Likely To Be Adversely Affected in the Action Area**

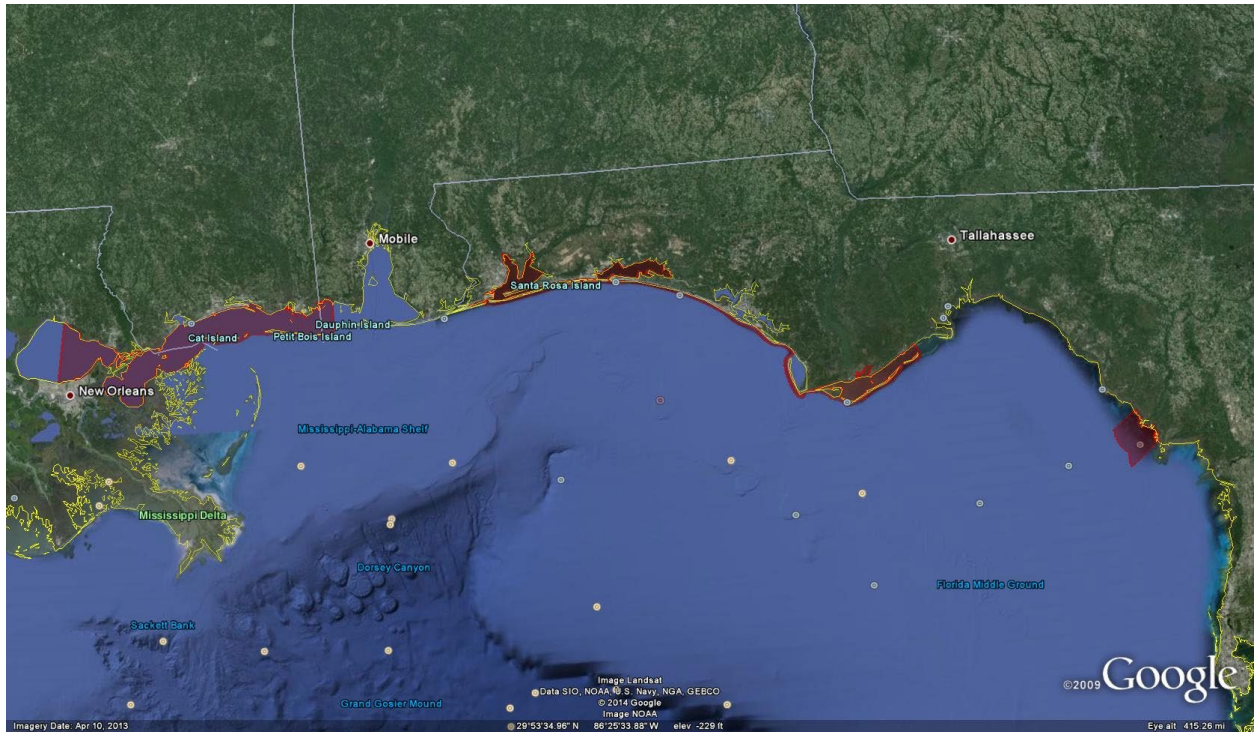
The term “critical habitat” is defined in Section 3(5)(A) of the ESA as (i) the specific areas within the geographic area occupied by a species, at the time it is listed in accordance with the Act, on which are found those physical or biological features (1) essential to the conservation of the species and (2) that may require special management considerations or protection; and (ii) specific areas outside the geographic area occupied by a species at the time it is listed, upon a determination that such areas are essential for the conservation of the species. “Conservation” is defined in Section 3(3) of the ESA as “...the use of all methods and procedures that are necessary to bring any endangered or threatened species to the point at which listing under the ESA is no longer necessary.”

The proposed action includes four low relief artificial reef site (FH-8, FH-9, FH-10, FH-11) located within the boundary of GSCH (Lake Pontchartrain – Mississippi Sound – Unit 8). We have determined that the proposed action is likely to adversely affect this designated critical habitat. Effects of the action on Gulf sturgeon critical habitat are discussed in Section 5.

#### **3.3.1 Status of Gulf Sturgeon Critical Habitat**

NMFS and USFWS jointly designated GSCH on April 18, 2003 (*see*, 50 CFR 226.214). The agencies designated 7 riverine areas (Units 1-7) and 7 estuarine/marine areas (Units 8-14) as critical habitat based on the physical and biological features that support the species. Critical habitat units encompass a total of 2,783 river kilometers (rkm) and 6,042 square kilometers (km<sup>2</sup>) of estuarine and marine habitats (Figure 8; Table 6). NMFS’s jurisdiction encompasses the 7 units in marine and estuarine waters (Units 8-14), though NMFS’s consultation responsibilities for projects in estuarine waters are limited to specific action agencies (Table 7).





**Figure 8. Gulf sturgeon critical habitat in estuarine and marine waters (Units 8-14) (©2014 Google)**

**Table 6. Approximate Area of the Estuarine and Marine Critical Habitat Units for Gulf Sturgeon**

Critical Habitat Unit	State	Km <sup>2</sup>
8. Lake Borgne	Louisiana/Mississippi/Alabama	718
Little Lake		8
Lake Pontchartrain		763
Lake St. Catherine		26
The Rigolets		13
Mississippi Sound		1,879
MS nearshore Gulf		160
9. Pensacola Bay	Florida	381
10. Santa Rosa Sound	Florida	102
11. Near Shore Gulf of Mexico	Florida	442
12. Choctawhatchee Bay	Florida	321
13. Apalachicola Bay	Florida	683
14. Suwannee Sound	Florida	546
Total		6,042

**Table 7. Consultation Responsibility for Projects in Estuarine Waters**

Lead Action Agency	NMFS	USFWS
Department of Transportation		X
U.S. Environmental Protection Agency		X
U.S. Coast Guard		X
Federal Emergency Management Agency		X
Department of Defense (DoD)	X	
U.S. Army Corps of Engineers	X	
Minerals Management Service (now Bureau of Ocean Energy Management)	X	
Other	X	

Gulf sturgeon use rivers for spawning, larval and juvenile feeding, adult resting and staging, and to move between the areas that support these components. Gulf sturgeon use the lower riverine, estuarine, and marine environment during winter months primarily for feeding and for inter-river migrations. Within Florida estuaries, Gulf sturgeon are typically found in waters 2-4 m deep and use depths outside this range less than expected based on availability (Fox et al. 2002a). Further, the 2-4-m deep habitats where Gulf sturgeon are typically found have sediments with a high percentage (> 80%) of sand (Fox et al. 2002a). Gulf sturgeon in Mississippi estuaries appear to occupy habitats with lower percentages of sand (typically < 75% sand) but similar depth ranges (Michael Anders, USM, unpublished data). Adult sturgeon appear to spend extended periods of time in specific areas of the estuary and then travel relatively quickly to other areas where they again spend extended amounts of time (Edwards et al. 2007; Edwards et al. 2003). Sulak et al. (2012) believe Gulf sturgeon feed continuously during these periods which may last for 1-3 months. Additionally, it appears that there may be certain areas where Gulf sturgeon concentrate. USFWS discovered nearshore areas of concentrated feeding activity for adults from multiple riverine systems in the waters near Tyndall Air Force Base/Panama City Beach, Florida, and

waters from Perdido, Florida, to Gulf Shores, Alabama (USFWS 2004; USFWS 2005; USFWS 2006; USFWS 2007). Estuaries and bays adjacent to riverine areas provide unobstructed passage of sturgeon from feeding areas to spawning grounds.

#### *Physical and Biological Features of Critical Habitat*

Critical habitat determinations focus on identifying physical and biological features (primary constituent elements [PCEs]) that are essential to the conservation of the species (50 CFR 424.12). NMFS and USFWS identified 7 habitat features essential for the conservation of Gulf sturgeon. Four of these features are found in the marine and estuarine units of critical habitat:

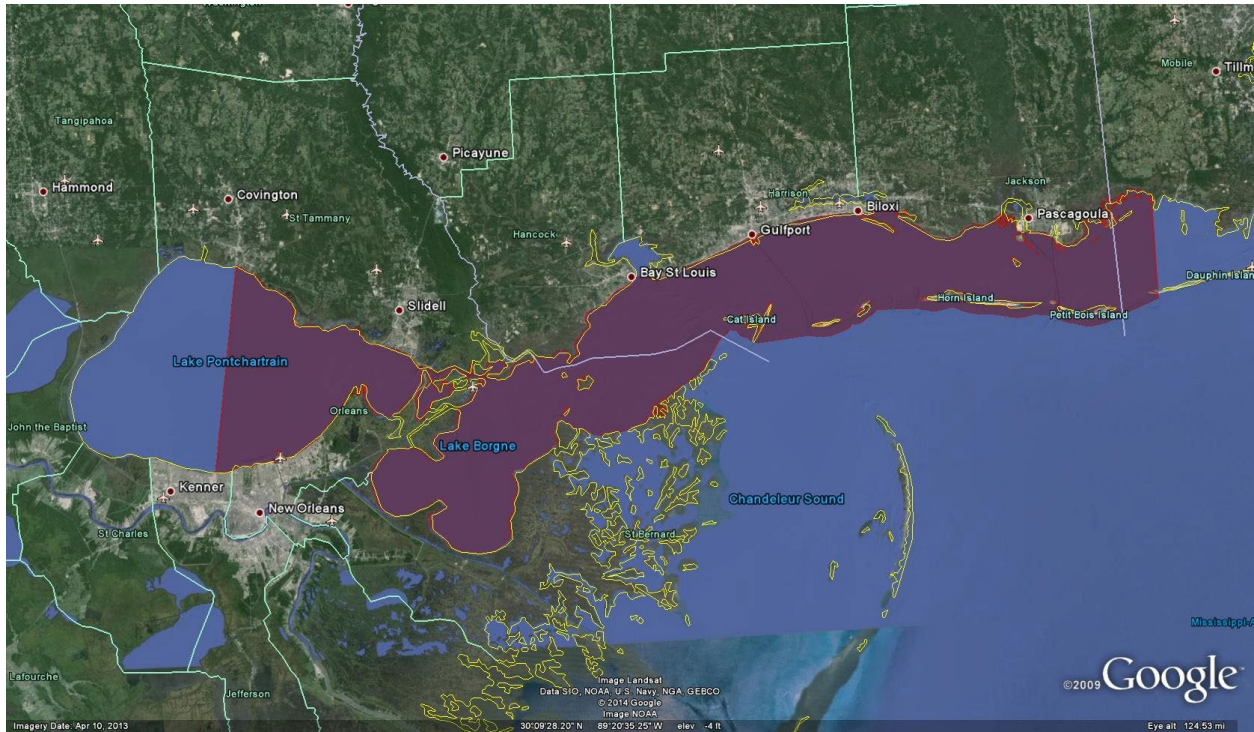
1. Abundant prey items, such as detritus, aquatic insects, worms, and/ or mollusks, within riverine habitats for larval and juvenile life stages; and abundant prey items, such as amphipods, lancelets, polychaetes, gastropods, ghost shrimp, isopods, mollusks and/or crustaceans, within estuarine and marine habitats and substrates for subadult and adult life stages;
2. Water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages;
3. Sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and
4. Safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (e.g., a river unobstructed by any permanent structure, or a dammed river that still allows for passage)

As stated in the final rule designating Gulf sturgeon critical habitat, the following activities, among others, when authorized, funded, or carried out by a federal agency, may destroy or adversely modify designated critical habitat (68 FR 13399):

- (1) Actions that would appreciably reduce the abundance of estuarine and marine prey for juvenile and adult Gulf sturgeon, within a designated critical habitat unit, such as dredging, dredged material disposal, channelization, in-stream mining; and land uses that cause excessive turbidity or sedimentation;
- (2) Actions that would alter water quality within a designated critical habitat unit, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, such that it is appreciably impaired for normal Gulf sturgeon behavior, reproduction, growth, or viability, such as dredging, dredged material disposal, channelization, impoundment, in-stream mining, water diversion, dam operations, land uses that cause excessive turbidity, and release of chemicals, biological pollutants, or heated effluents into surface water or connected groundwater via point sources or dispersed non-point sources;
- (3) Actions that would alter sediment quality within a designated critical habitat unit such that it is appreciably impaired for normal Gulf sturgeon behavior, reproduction, growth, or viability, such as dredged material disposal, channelization, impoundment, in-stream mining, land uses that cause excessive sedimentation, and release of chemical or biological pollutants that accumulate in sediments; and

- (4) Actions that would obstruct migratory pathways within and between adjacent riverine, estuarine, and marine critical habitat units, such as dams, dredging, point-source-pollutant discharges, and other physical or chemical alterations of channels and passes that restrict Gulf sturgeon movement.

Elements of the proposed action will occur in Gulf sturgeon critical habitat Unit 8. The placement of artificial reef material at FH sites 8-11 may affect critical habitat. Hence, NMFS believes designated critical habitat for Gulf sturgeon may be adversely affected by the proposed action. Unit 8 encompasses Lake Pontchartrain east of the Lake Pontchartrain Causeway, all of Little Lake, The Rigolets, Lake St. Catherine, Lake Borgne, including Heron Bay, the Mississippi Sound, and the Gulf of Mexico (Figure 9). Critical habitat follows the shorelines around the perimeters of each included lake. The Mississippi Sound includes adjacent open bays including Pascagoula Bay, Point aux Chenes Bay, Grand Bay, Sandy Bay, and barrier island passes, including Ship Island Pass, Dog Keys Pass, Horn Island Pass, and Petit Bois Pass. The northern boundary of the Mississippi Sound is the shoreline of the mainland between Heron Bay Point, Mississippi and Point aux Pins, Alabama. Critical habitat excludes St. Louis Bay, north of the railroad bridge across its mouth; Biloxi Bay, north of the U.S. Highway 90 bridge; and Back Bay of Biloxi. The southern boundary follows along the broken shoreline of Lake Borgne created by low swamp islands from Malheureux Point to Isleau Pitre. From the northeast point of Isleau Pitre, the boundary continues in a straight north-northeast line to the point 1 nautical mile (nm) (1.9 km) seaward of the western most extremity of Cat Island (30°13'N, 89°10'W). The southern boundary continues 1 nm (1.9 km) offshore of the barrier islands and offshore of the International Regulations for Preventing Collisions at Sea 1972 (72 COLREGS) lines at barrier island passes (defined at 33 CFR 80.815(c), (d) and (e)) to the eastern boundary. Between Cat Island and Ship Island there is no 72 COLREGS line. Therefore, that section of the unit southern boundary is defined as 1 nm (1.9 km) offshore of a straight line drawn from the southern tip of Cat Island to the western tip of Ship Island. The eastern boundary is the line of longitude 88°18.8'W from its intersection with the shore (Point aux Pins) to its intersection with the southern boundary. The lateral extent of Unit 8 is the mean high water line on each shoreline of the included water bodies or the entrance to rivers, bayous, and creeks. Major shipping channels in this unit, as identified on standard navigation charts and marked by buoys, are excluded under section 4(b)(2) of the ESA. 50 CFR 226.214(h).



**Figure 9. Gulf sturgeon critical habitat Unit 8.**

The Pearl River and its distributaries flow into The Rigolets, Little Lake, and Lake Borgne, which is the western extension of Mississippi Sound. The Rigolets connect Lake Pontchartrain and Lake St. Catherine with Little Lake and Lake Borgne. The Pascagoula River and its distributaries flow into Pascagoula Bay and Mississippi Sound. This unit provides juvenile, subadult, and adult feeding, resting, and passage habitat for Gulf sturgeon from the Pascagoula and the Pearl River subpopulations. One or both of these subpopulations have been documented by tagging data, historic sightings, and incidental captures as using Pascagoula Bay, The Rigolets, the eastern half of Lake Pontchartrain, Little Lake, Lake St. Catherine, Lake Borgne, Mississippi Sound, within 1 nm (1.9 km) of the nearshore Gulf of Mexico adjacent to the barrier islands and within the passes (Davis et al. 1970; Morrow et al. 1998; Reynolds 1993; Rogillio 1993; Rogillio et al. 2001; Rogillio et al. 2007; Ross et al. 2001; Ross et al. 2009) (F. Parauka, USFWS, pers. comm. to S. Bolden, NMFS, December 2, 2002). Substrate in these areas range from sand to silt, all of which contains known Gulf sturgeon prey items. The Rigolets is an 11.3 km (7 mi)-long and about 0.6 km (0.4 mi)-wide passage connecting Lake Pontchartrain and Lake Borgne. This brackish water area is used by adult Gulf sturgeon as a staging area for osmoregulation and for passage to and from wintering areas (Rogillio et al. 2001). Lake St. Catherine is a relatively shallow lake with depths averaging approximately 1.2 m (4 ft), connected to The Rigolets by Sawmill Pass. Bottom sediments in Sawmill Pass are primarily silt; bottom sediments in Lake St. Catherine are composed of silt and sand (Barrett 1971). Incidental catches of Gulf sturgeon are documented from Lake St. Catherine and Sawmill Pass (Reynolds 1993) (H. Rogillio, Louisiana Department of Wildlife and Fisheries, pers. comm. to S. Bolden, NMFS, December 2, 2002). Based on the proximity of Little Lake, Lake St. Catherine, and Sawmill Pass to The Rigolets and Pearl River, we believe these areas are also used for staging and feeding and, therefore, were included with The Rigolets as designated critical habitat.

The Mississippi Sound is separated from the Gulf of Mexico by a chain of barrier islands, including Cat, Ship, Horn, and Petit Bois Islands. Natural depths of 3.7-5.5 m (12-18 ft) are found throughout the Sound and a channel 3.7 m (12 ft) deep has been dredged where necessary from Mobile Bay to New Orleans. Incidental captures and studies confirm that both Pearl River and Pascagoula River adult Gulf sturgeon winter in the Mississippi Sound, particularly around barrier islands and barrier islands passes (Reynolds 1993; Rogillio et al. 2001; Ross et al. 2001). Pascagoula Bay is adjacent to the Mississippi Sound. Gulf sturgeon exiting the Pascagoula River move both east and west, with telemetry locations as far east as Dauphin Island and as far west as Cat Island and the entrance to Lake Pontchartrain (Ross et al. 2001). Tagged Gulf sturgeon from the Pearl River subpopulation have been located between Cat Island, Ship Island, Horn Island, and east of Petit Bois Islands to the Alabama State line (Rogillio et al. 2001). Gulf sturgeon have also been documented within 1 nm (1.9 km) off the barrier islands of Mississippi Sound. Thus, the area 1 nm (1.9 km) offshore of the barrier islands of Mississippi Sound is included in Unit 8.

Habitat used by Gulf sturgeon in the vicinity of the barrier islands is 1.9-5.9 m (6.2-19.4 ft) deep (average 4.2 m [13.8 ft]), with clean sand substrata (Heise et al. 1999; Rogillio et al. 2001; Ross et al. 2001). Preliminary data from substrate samples taken in the barrier island areas indicate that all samples contained lancelets (Ross et al. 2001). Inshore locations where Gulf sturgeon were located (Deer Island, Round Island) were 1.9-2.8 m (6.2-9.2 ft) deep and all had mud (mostly silt and clay) substrata (Heise et al. 1999), typical of substrates supporting known Gulf sturgeon prey.

#### *Status of Critical Habitat Unit 8*

Activities associated with coastal development have been and continue to be the primary threat to marine and estuarine units of GSCH. These activities generally include dredge and fill projects, freshwater withdrawals, and storm water drainage systems. Although many coastal development activities are currently regulated, some permitted direct and/or indirect damage to habitat from increased urbanization still occurs and is expected to continue in the future.

Unit 8 is impacted by a number of activities including dredging, shoreline armoring, installation of breakwaters, and construction of docks, piers, marinas, and artificial reefs. These types of projects are not expected to adversely affect the essential features of Unit 8, and any impacts from these projects are expected to be temporary, and therefore, insignificant, or extremely unlikely to occur.

The settling of contaminants from the DWH oil spill has the potential to adversely affect PCEs of GSCH. According to an analysis conducted by NMFS on the DWH spill oiling legacy (M. Press, NMFS, memo to D. Bernhart, NMFS, December 8, 2014), indirect impacts to water quality could occur if dredging disturbed a submerged oil mats (SOMs) remnant of the DWH spill. Contributors to Operational Science Advisory Team (OSAT III) report have stated that while there is a possibility that dredging activities could re-suspend oil into the water column in certain areas, the likelihood of this happening is low. Additionally, an analysis of the matrix of material (oil plus sand) stranded in mats revealed that SOMs were composed mostly of sand: 83.2%-90.6% sand and 9.4%-16.8% oil (OSAT II). The effects of suffocation of infaunal organisms and toxicity of substrate would impact potential foraging areas within GSCH through the



displacement and/or reduction of prey items; however, indirect impacts to sediment quality and direct effects to prey abundance would be insignificant. The likelihood of any re-suspended DWH oil being toxic is low, and should also not have measurable effects on water quality (or on listed species directly) (W. Bryant, OSAT III Science Team Lead, pers. comm. to M. Press, NMFS, July 31, 2014). NMFS concluded that the dredging activities were not likely to adversely affect GSCH.

## **4 ENVIRONMENTAL BASELINE**

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This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and the ecosystem, within the action area. The environmental baseline describes the status of Kemp's ridley, loggerhead, leatherback and green sea turtles, as well as GSCH's health within the action area 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 the 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 current state of a species and its designated critical habitat is important because some species and critical habitat features will commonly exhibit, or be more susceptible to, adverse responses to stressors than they would be to other species or in other areas, or may have been exposed to unique or disproportionate stresses. These localized stress responses or stressed baseline conditions may increase the severity of the adverse effects expected from the proposed action.

### **4.1 Status of Species and Critical Habitat within the Action Area**

As discussed above, this Opinion focuses on an activity occurring in Unit 8 of Gulf sturgeon designated critical habitat.

#### *Sea Turtles*

The species of sea turtles covered under this Opinion that occur in the action area (green [NA and SA DPS]), Kemp's ridley, loggerhead NWA DPS, and leatherback sea turtles) are all highly migratory. The status of these species in the action area, as well as the threats to these species, are supported by the species accounts in Section 3 (Status of the Species).

#### *Gulf Sturgeon Critical Habitat*

The project sites are located in the Mississippi Sound and the Gulf of Mexico off the coast of Mississippi. Sites FH-8, FH-9, FH-10 and FH-11 are located in GSCH (Unit 8 – Lake

Pontchartrain – Mississippi Sound). The status of critical habitat in the action area, as well as the threats to this habitat, are best reflected in the current status in Section 3 of this Opinion (Status of Critical Habitat).

## **4.2 Factors Affecting Sea Turtles within the Action Area**

### **Federal Actions**

We have undertaken a number of Section 7 consultations to address the effects of federally-permitted dredging and other federal actions on threatened and endangered sea turtle 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 sea turtles. The summary below of federal actions and the effects these actions have had on sea turtles includes only those federal actions in the action areas which have already concluded or are currently undergoing formal Section 7 consultation.

### **Existing Artificial Reef Sites**

Artificial reef structures exist in the area and are the subject of ESA consultation between the USACE and NMFS. Low-relief artificial reef sites designated as Fish Haven 3 (FH-3), FH-4, FH-5, FH-6, FH-12, FH-13, were the subject of NMFS consultation number SERO-2019-00045. High-relief artificial reef sites FH-1, FH-2 and FH-7 and low-relief artificial reef sites FH-8, FH-9, FH-10 and FH-11 were the subject of NMFS consultation SERO-2019-03400. Mississippi Reef Zone 1 (MRZ1) is adjacent to, and covers the area between, FH1 and FH-2. Mississippi Reef Zone 2 (MRZ2) is adjacent to, and southeast of, FH-13.

### **Federal Dredging Activity**

Marine dredging vessels are common within U.S. coastal waters. Although the underwater noises from dredge vessels are typically continuous in duration (for periods of days or weeks at a time) and strongest at low frequencies, they are not believed to have any long-term effect on sea turtles. Still, the construction and maintenance of federal navigation channels and dredging in sand mining sites (borrow areas) have been identified as sources of sea turtle mortality. Hopper dredges in the dredging mode are capable of moving relatively quickly compared to sea turtle swimming speed and can thus overtake, entrain, and kill sea turtles as the suction draghead(s) of the advancing dredge overtakes the resting or swimming turtle. Entrained sea turtles rarely survive. The Federal dredging program in closest proximity to the action area is the channel improvements to Gulfport Harbor (SERO-2019-02044). The Federal navigation channel is located just to the west of the action area. The SERO-2019-02044 Opinion included an Incidental Take Statement (ITS) and determined that hopper dredging during the proposed actions would not jeopardize any species of green (NA and SA DPS), Kemp's ridley, loggerhead NWA DPS, or leatherback sea turtles or other ESA-listed species, and would not destroy or adversely modify critical habitat of any listed species.

To reduce lethal 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. Relocation trawling has been successful and routinely moves sea turtles in the Gulf of Mexico.



### **Federal Vessel Activity**

Watercraft are the greatest contributors to overall noise in the sea and have the potential to interact with sea turtles through 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. Potential sources of adverse effects from federal vessel operations in the action area include operations of the BOEM, Federal Energy Regulatory Commission (FERC), USCG, NOAA, and USACE. The Gulfport, MS navigational channel is located just to the west of the action area and improvements to the channel for vessel traffic was addressed in NMFS's SERO-2019-02044 Opinion (referenced above).

We have also conducted Section 7 consultations related to energy projects in the Gulf of Mexico (BOEM, FERC, and USCG) to implement conservation measures for vessel operations. 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. At the present time, they present the potential for some level of interaction.

Operations of vessels by other federal agencies within the action area (NOAA, BOEM) may adversely affect sea turtles. Yet, the in-water activities of those agencies are limited in scope, as they operate a limited number of vessels or are engaged in research/operational activities that are unlikely to contribute a large amount of risk.

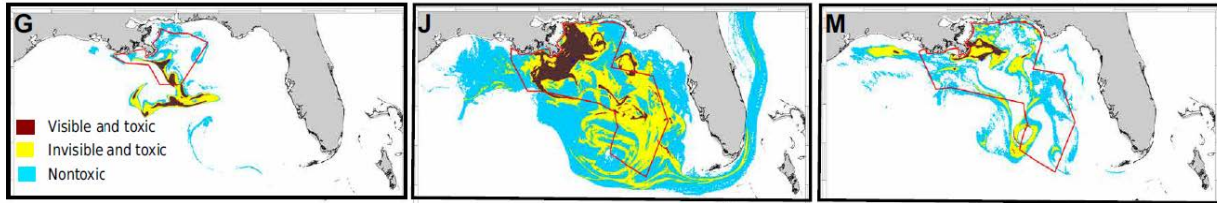
### **Oil and Gas Exploration and Extraction**

Oil and gas exploration, production, and development in the Gulf of Mexico Federally regulated by the BOEM and the USEPA are the subject of a NMFS's programmatic Biological Opinion under the NMFS consultation number FPR-2017-9234. These activities are expected to result in some sublethal effects to ESA-listed sea turtles, including impacts associated with pile driving for, or the explosive removal of, offshore structures, seismic exploration, marine debris, and oil spills. The primary causes of mortality are related to vessel strikes, oil spills and marine debris.

#### *Impact of DWH Oil Spill on Status of Sea Turtles*

On April 20, 2010, while working on an exploratory well approximately 50 miles offshore Louisiana, the semi-submersible drilling rig DWH experienced an explosion and fire. The rig subsequently sank and oil and natural gas began leaking into the Gulf of Mexico. Oil flowed for 86 days, until the well was finally capped on July 15, 2010. Millions of barrels of oil were released into the Gulf. Additionally, approximately 1.84 million gallons of chemical dispersant was applied both subsurface and on the surface to attempt to break down the oil.

The DWH event and associated response activities (e.g., skimming, burning, and application of dispersants) have resulted in adverse effects on ESA-listed sea turtles. The maps below show the spread of the DWH spill and the areas affected, which includes the action area. The effects of the DWH spill on the ESA-listed sea turtles and Gulf sturgeon critical habitat was discussed in Section 3, above.



**Figure 10.** The spread of the impacts from the DWH spill; G from 15 May 2010, J from 18 June 2010, M from 2 July 2010 (Berenshtein et al. 2020).

## ESA Permits

Sea turtles are the focus of research activities authorized by Section 10 permits under the ESA. 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. Authorized activities range from photographing, weighing, and tagging sea turtles incidentally taken in fisheries, to blood sampling, tissue sampling (biopsy), and performing laparoscopy on intentionally captured sea turtles. The number of authorized takes varies widely depending on the research and species involved, but may involve the taking of hundreds of sea turtles annually. Most takes authorized under these permits are expected to be (and are) nonlethal. Before any research permit is issued, the proposal must be reviewed under the permit regulations. In addition, since issuance of the permit is a federal activity, our issuance of the permit must also 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 the destruction or adverse modification of its critical habitat.

## Fisheries

Threatened and endangered sea turtles are adversely affected by fishing gears used throughout the continental shelf of the action area. Gillnet, pelagic and bottom longline, other types of hook-and-line gear, trawl, and pot fisheries have all been documented as interacting with sea turtles. The Gulf of Mexico Fishery Management Council develops and amends Fishery Management Plans (FMPs) for various fishery resources within the Gulf of Mexico and NMFS consults on these FMPs through the Section 7 consultation process. The FMPs and their amendments applicable to the range of the action area include Coastal Migratory Pelagic FMP, Reef Fish FMP, and Shrimp FMP. Some of these consultations resulted in subsequent rulemaking to reduce the impacts of the specific fisheries on sea turtle populations. Examples include additional monitoring of and TED requirements in the southeast U.S. shrimp fisheries, as well as gear limitations and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality in Atlantic highly migratory species fisheries and reef fish fisheries. All Opinions had an ITS and determined that fishing activities, as considered (i.e., with conservation requirements) would not jeopardize any species of sea turtles or other listed species, or destroy or adversely modify critical habitat of any listed species.

## State or Private Actions

### 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*

Trawls that operate in the action area may adversely affect sea turtles. On December 16, 2016, we published a notice of availability of our DEIS (EIS No. 20160294; 81 FR 91169) as well as a proposed rule (81 FR 91097) in the *Federal Register* to address incidental bycatch and mortality of sea turtles in the Southeastern U.S. shrimp fisheries. The proposed rule would have revoked the alternative tow time restrictions for skimmer trawls, pusher-head trawls, and wing nets (butterfly trawls) at 50 CFR 223.206(d)(2)(ii)(A)(3), and require those vessels to use TEDs designed to exclude small turtles while fishing. On December 20, 2019 (84 FR 70048), we published a final rule that requires all skimmer trawls 40 feet and greater in length to use TEDs designed to exclude small sea turtles in their nets effective August 1, 2021.

### **Recreational Fishing**

Recreational fishing as regulated by Mississippi and Louisiana can affect protected species or their habitats within the action area. Recreational fishing from private vessels may occur in the action area. 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, piers, 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 (TEWG) reports (Turtle Expert Working Group 1998a; Turtle Expert Working Group 2000).

### **Vessel Traffic**

Commercial traffic and recreational boating pursuits can have adverse effects on sea turtles via propeller and boat strike damage. Data show that vessel traffic is one cause of sea turtle mortality (Environment Australia 2003; Hazel and Gyuris 2006; Lutcavage et al. 1997). The STSSN data from 2007 – 2016 for the Mississippi Zones 11 and 12 (which includes the action area) includes 102 records of vessel interactions with sea turtles, which were all fatal.<sup>9</sup> Data indicate that 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.

Data show that vessel traffic is one cause of sea turtle mortality (Environment Australia 2003; Hazel and Gyuris 2006; Lutcavage et al. 1997). Stranding data for the Gulf of Mexico coast

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<sup>9</sup> Sea Turtle Stranding and Salvage Network (STSSN). <https://www.fisheries.noaa.gov/national/marine-life-distress/sea-turtle-stranding-and-salvage-network>

show that vessel-related injuries are noted in stranded sea turtles. 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.

### **Coastal Development**

Beachfront development, lighting, and beach erosion control all are ongoing activities along the Mississippi and Louisiana coastlines, including the barrier islands near and within the action area. ESA consultations within or near the action area include pier repairs and/or removals on Ship Island (SERO-2019-00536) and Horn Island SERO-2019-00471) and Round Island breakwater marker and sign repair. 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. Coastal counties are adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting.

### **Climate Change**

As discussed in Section 3.2.1.1 of this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. Potential effects commonly mentioned include changes in sea temperatures and salinity (due to melting ice and increased rainfall), ocean currents, storm frequency and weather patterns, and ocean acidification. These changes have the potential to affect species behavior and ecology including migration, foraging, reproduction (e.g., success), and distribution. For example, sea turtles currently range from temperate to tropical waters. A change in water temperature could result in a shift or modification of range. Climate change may also affect marine forage species, either negatively or positively (the exact effects for the marine food web upon which sea turtles rely is unclear, and may vary between species). It may also affect migratory behavior (e.g., timing, length of stay at certain locations). These types of changes could have implications for sea turtle recovery within the action area.

With regard to the action area, global climate change may affect the timing and extent of population movements and their range, distribution, species composition of prey, and the range and abundance of competitors and predators. Changes in distribution including displacement from ideal habitats, decline in fitness of individuals, population size due to the potential loss of foraging opportunities, abundance, migration, community structure, susceptibility to disease and contaminants, and reproductive success are all possible impacts that may occur as the result of climate change. Still, more information is needed to better determine the full and entire suite of impacts of climate change on sea turtles and specific predictions regarding impacts in the action area are not currently possible.

### **Other Potential Sources of Impacts to the Environmental Baseline**

#### **Stochastic events**

Stochastic (i.e., random) events, such as hurricanes, occur in the northern Gulf of Mexico 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. Other stochastic events, such as a winter cold snap, can injure or kill sea turtles.

### **Marine Pollution and Environmental Contamination**

Coastal runoff, marina and dock construction, dredging, aquaculture, increased under water noise and boat traffic can degrade marine habitats used by sea turtles (Colburn et al. 1996) and negatively impact nearshore habitats, including the action area. Fueling facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive estuarine and coastal habitats. Although these contaminant concentrations are unknown in the action area, the sea turtles analyzed in this Opinion travel within near shore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles.

The Gulf of Mexico is an area of high-density offshore oil extraction with chronic, low-level spills and occasional massive spills (e.g., the DWH oil spill event). As discussed above, 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 and McGehee 1982; Lutcavage et al. 1997; Witherington 1999).

The accumulation of organic contaminants and trace metals has been studied in loggerhead, green, and leatherback sea turtles (Aguirre et al. 1994; Caurant et al. 1999; Corsolini et al. 2000) (Mckenzie et al. 1999). Omnivorous loggerhead sea 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. (Sakai et al. 1995) found the presence of metal residues occurring in loggerhead sea turtle organs and eggs. (Storelli et al. 1998) analyzed tissues from 12 loggerhead sea turtles stranded along the Adriatic Sea (Italy) and found that characteristically, mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991). No information on detrimental threshold concentrations is available, and little is known about the consequences of exposure of organochlorine compounds to sea turtles. Research is needed on the short- and long-term health and fecundity effects of chlorobiphenyl, organochlorine, and heavy metal accumulation in sea turtles.

### **Conservation and Recovery Actions Benefitting Sea Turtles**

NMFS has implemented a number of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for Atlantic Highly Migratory Species (HMS) and Gulf of Mexico reef fish fisheries, and TED requirements for the southeastern shrimp fisheries. TEDs and other bycatch reduction device requirements may reduce sea turtle bycatch in Southeast trawl fisheries (Atlantic Sturgeon Status Review Team 2007). NMFS has required the use of TEDs in southeast United States shrimp trawls since 1989 and in summer flounder trawls in the mid-Atlantic area (south of Cape Charles, Virginia) since 1992 to reduce the potential for incidental mortality of

sea turtles in commercial trawl fisheries. In addition to regulations, outreach programs have been established and data on sea turtle interactions with recreational fisheries has been collected through the Marine Recreational Fishery Statistical Survey (MRFSS)/Marine Recreational Information Program.

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

We, along with cooperating states, 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 eighteen-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.

In response to the growing awareness of recreational fishery impacts on sea turtles, in 2006 the Marine Recreational Fishery Statistics Survey added a survey question regarding sea turtle interactions within recreational fisheries. We are exploring potential revisions to Marine Recreational Information Program to quantify recreational encounters with sea turtles on a permanent basis.

#### **4.3 Factors Affecting Critical Habitat within the Action Area**

The April 2003 joint designation of GSCH by NMFS and USFWS will benefit the species' habitat, primarily through the ESA section 7 consultation process. When critical habitat is designated, other federal agencies are required to consult with NMFS on actions they carry out, fund, or authorize, to ensure that their actions will not destroy or adversely modify critical habitat. In this way, critical habitat designation protects physical and biological features that are necessary for the conservation of the species. Designation of critical habitat may also enhance awareness within federal agencies and the general public of the importance of Gulf sturgeon habitat and the need for special management considerations.

##### *DWH Legacy Effects on GSCH*

The settling of contaminants from the DWH oil spill has the potential to adversely affect PCEs of GSCH. According to an analysis conducted by NMFS on the DWH spill oiling legacy (M. Press, NMFS, memo to D. Bernhart, NMFS, December 8, 2014), indirect impacts to water quality could occur if dredging disturbed a submerged oil mats (SOMs) remnant of the DWH spill. Contributors to Operational Science Advisory Team (OSAT III) report have stated that while there is a possibility that dredging activities could re-suspend oil into the water column in certain areas, the likelihood of this happening is low. Additionally, an analysis of the matrix of material (oil plus sand) stranded in mats revealed that SOMs were composed mostly of sand: 83.2%-90.6% sand and 9.4%-16.8% oil (OSAT II). The effects of suffocation of infaunal organisms and toxicity of substrate would impact potential foraging areas within GSCH through the

displacement and/or reduction of prey items; however, indirect impacts to sediment quality and direct effects to prey abundance would be insignificant. The likelihood of any re-suspended DWH oil being toxic is low, and should also not have measurable effects on water quality (or on listed species directly) (W. Bryant, OSAT III Science Team Lead, pers. comm. to M. Press, NMFS, July 31, 2014). NMFS concluded that the dredging activities were not likely to adversely affect GSCH.

#### *Artificial Reef Effects on GSCH*

NMFS has previously reviewed the effects of the placement of artificial reef material in GSCH. A 2005 Opinion (SER-2005-06196) addresses the deployment of concrete bridge rubble at 4 disposal sites in the Mississippi Sound located in GSCH Unit 8, and impacted 50 ac. NMFS concluded that the migratory pathways, sediment quality, and prey abundance PCEs would be adversely affected by the project but not to the extent that would reduce the critical habitat's ability to support Gulf sturgeon conservation, and therefore not likely to destroy or adversely modify designated GSCH.

A 2020 Opinion (SERO-2018-00252, SER 2018-19361) also addressed the deployment of low-relief materials of opportunity at a 62-ac reef site located off of Cat Island in the Mississippi Sound in GSCH Unit 8. NMFS concluded that the sediment quality PCE would be adversely affected by the project but not to the extent that would reduce the critical habitat's ability to support Gulf sturgeon conservation, and therefore not likely to destroy or adversely modify designated GSCH.

## **5 EFFECTS OF THE ACTION ON SPECIES AND CRITICAL HABITAT**

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In this section of our Opinion, we assess the effects of the proposed action that are likely to adversely affect ESA-listed species and designated critical habitat. As noted above, we believe the ESA-listed Kemp's ridley, loggerhead [NWA DPS], green [NA and SA DPSs], and leatherback sea turtles and designated critical habitat for Gulf sturgeon (GSCH Unit 8) are likely to be adversely affected. This section describes the potential adverse effects to these species and designated critical habitat. The analysis in this section forms the foundation for our jeopardy analysis in Section 7.0. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Data are limited, so we are often forced to make assumptions to overcome the limits in our knowledge. Sometimes, the best available information may include a range of values for a particular aspect under consideration, or different analytical approaches may be applied to the same data set. In those cases, the uncertainty is resolved in favor of the species (House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)). NMFS generally selects the value or makes the assumption 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.

Effects of the action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time

and may include consequences occurring outside the immediate area involved in the action (50 CFR 402.02).

As discussed in Section 3, 5 species of ESA-listed sea turtles may be adversely affected by the proposed action. In addition, because 4 of the FH sites are located within the boundary of Gulf sturgeon designated critical habitat, we believe the proposed action is likely to adversely affect 1 of the 4 physical and biological features of designated critical habitat as described in Section 3.3, as further discussed in Section 5.2.

### **5.1 Potential Effects to ESA-Listed Sea Turtles from Entanglement**

NMFS believes that the presence of high-relief artificial reef material is likely to adversely affect Kemp's ridley sea turtle, loggerhead sea turtle (NWA DPS), green sea turtle (NA and SA DPSs), and leatherback sea turtle. High-relief artificial reef material specifically refers to vessels, aircrafts, decommissioned oil rigs, bridge spans, metal towers, or similar material that extends 7 ft or more from the seafloor and that has a footprint greater than 200 ft<sup>2</sup> (individually or collectively), excluding prefabricated artificial reef modules.

#### *Approach to Assessment*

We began our analysis of the effects of the action by first reviewing what activities associated with the proposed action are likely to adversely affect sea turtles in the action area (i.e., what the proposed action stressors are). We next review the range of responses to an individual's exposure to that stressor, and the factors affecting the likelihood, frequency, and severity of exposure. Afterwards, our focus shifts to evaluating and quantifying exposure. We estimate the number of individuals of each species likely to be exposed and the likely fate of those animals.

Since the proposed action will deploy high-relief material (vessels, aircrafts, decommissioned oil rigs, bridge spans, metal towers and similar material), we anticipate adverse effects on Kemp's ridley, loggerhead (NWA DPS), green (NA and SA DPSs), and leatherback sea turtles from entanglement and drowning in monofilament and other entangling gear that accumulates on that type of reef material. In general, due to the absence of monofilament immediately following an artificial reef deployment, we expect the risk of entanglement to be extremely low for some period of years. However, as time passes and monofilament line accumulates, the probability of an entanglement event increases. Also, the longer the accumulated line is present, the greater the chance that a sea turtle will encounter it. The rate of monofilament accumulation and the time it takes to reach the level where we might anticipate an entanglement-related mortality likely varies significantly due to the factors previously mentioned. As time passes, the integrity of the high-relief material will become compromised and the structure may undergo significant and dramatic collapse. In some areas of the southeastern U.S., this process is facilitated by hurricane events. Regardless, over time, this will reduce the amount of vertical relief, but not eliminate the likelihood of monofilament accumulation. Therefore, the risk of an entanglement event persists, but perhaps at a somewhat lower level.

In some instances though, this collapse may increase the risk of entanglement. For example, as discussed in (Barnette 2017), intact vessels sunk as artificial reefs off South Florida may not present a high risk of entanglement initially, even with significant monofilament entanglement, as sea turtles are frequently observed at the sand/hull interface where there is little entangled line.



This potential preference may “shield” them from greater entanglement risk present on the deck and upper structures. Once the vessel collapses, however, the reduced relief of the vessel places entangled monofilament in closer proximity to the seabed and to sea turtles utilizing the material. The probability of entanglement could also remain fairly high or increase in areas that are not typically exposed to current that could otherwise abrade or help accumulate and incorporate entangled monofilament.

Based on best available information presented in (Barnette 2017) and STSSN data, we anticipate adult loggerhead and Kemp’s ridley sea turtles will be the sea turtle species primarily associated with entanglement events on high relief artificial reef material as a result of the proposed action. This is likely due to the species habitat preferences and other life history characteristics. Studies evaluating sea turtle dive profiles and depth distribution are limited and generally have focused on female sea turtles, likely due to the ease of tagging during nesting activities. While this is still useful, as it provides information on depth ranges where interesting female sea turtles may spend a significant amount of their time, it does not provide the full depth range in which all sea turtles may be exposed to entanglement risk on artificial reefs. For example, Houghton et al. (2002), while examining the diving depth profiles of two female loggerhead sea turtles during nesting, documented a maximum diving depth of 230 ft; though they noted the vast majority of the interesting interval was spent at depths less than 66 ft. While loggerheads have been documented diving to depths exceeding 760 ft (Sakamoto et al. 1990), other studies have demonstrated the majority of dives are occurring at much shallower depths. For instance, Arendt et al. (2012) documented most dives were conducted shallower than 160 ft, and were typically between 65-130 ft, when looking at male loggerhead sea turtles off the southeastern U.S. However, one of the authors of this study noted that one of the limitations about diving behavior is that a lot of the depths reflect where animals were captured and individual animal preferences, and do not reflect comprehensive diving behavior across the species as a whole (M. Arendt, SCDNR, pers. comm. with NMFS Biologist M. Barnette). While it may be useful to discount a depth below which we believe the threat of entanglement from monofilament on artificial reefs is unlikely to occur based on sea turtle diving behavior, the available literature is insufficient to support an adequately informed decision regarding the appropriate depth threshold. As a result, all complex, high-relief materials deployed as an artificial reef, regardless of depth, are included in our analysis.

Similarly, while it might make sense to scale the threat based on areas where we believe current or other oceanographic parameters, sea turtle densities, fishing patterns, artificial reef size, or other factors may decrease or increase the risk of entanglement from monofilament and other lines fouled on artificial reef material, the limited available information is insufficient to do so. Therefore, to be conservative, we will consider that all complex, high-relief materials deployed as artificial reefs will present similar entanglement risks to sea turtles over time, regardless of their location.

Barnette (2017) documents that a site submerged for more than 120 years appears to be still accumulating monofilament and result in sea turtle mortalities due to entanglement events. Given the remaining structure on that site, it is likely to persist for another 30 years (Barnette 2017). Therefore, for purposes of this analysis, we will use an effective lifespan of 150 years for vessels, decommissioned oil rigs, bridge spans, and other large metal structures.

Frequency of entanglement likely varies greatly by site due to numerous factors. As a result of limited information on the subject, however, it is not practical or feasible to examine these issues further. Barnette (2017) documents that several sites using vessels have had repeated instances of sea turtle entanglement over time, and there was documentation of one site with multiple entanglements. Although specific reasons for the number of entanglements at this reef site have not been identified, some artificial reefs appear to present a more significant threat of entanglement than others due to sea turtle habitat preference, migration corridors, reef structure or composition, or other environmental parameter (Barnette, 2017). The author also noted that evidence of sea turtle entanglement events is ephemeral, and the absence of evidence of entanglement should not be viewed as evidence that entanglements have not occurred. Perhaps some complex, high-relief artificial reefs will never result in a sea turtle mortality due to entanglement, but given the available information, we take a risk-averse approach and consider all vessels, decommissioned oil rigs, bridge spans, and other large metal structures deployed as artificial reefs similarly.

The lack of ongoing monitoring and the ephemeral nature of turtle entanglement evidence documented in Barnette (2017) (i.e., decomposition, current, predation, etc.) presents difficulties in estimating an annual take rate due to entanglement. For purposes of this analysis, based on the findings in Barnette (2017), our informed judgement, and taking a risk-averse approach, we will assume a 25-year delay of significant entanglement risk. After that point, we conservatively assume any high-relief artificial structure may result in 1 sea turtle mortality due to entanglement per year on a "mature" artificial reef site (i.e., a site that has accumulated sufficient line to present a lethal threat). Serious entanglement will effectively anchor a sea turtle to the artificial reef and prevent it from reaching the surface to breath, resulting in sea turtle mortality due to drowning (i.e., forced submergence). Numerous entanglement examples are documented in Barnette (2017). We consider this effect to be ongoing for the next 75 years for vessels, decommissioned oil rigs, bridge spans, and other large metal structures. After that point, we anticipate entanglement risk will be reduced on average due to material deterioration and subsidence. The entanglement risk over the next 50 years of the material's effective lifespan will result in one sea turtle mortality every 3 years. This translates to an estimated take of 92 sea turtles over 150 years resulting from the deployment of a single vessel, decommissioned oil rig, bridge span, or other large metal structures.

Assuming 1 structure (i.e., a vessel, decommissioned oil rig, bridge span, or other large metal structure) of high relief artificial reef material results in 0 sea turtle mortalities for the first 25 years, transitions to 1 sea turtle mortality each year for the next 75 years, and changes to 1 sea turtle mortality every 3 years for the following 50 years, we calculate the overall sea turtle take for the 150-year period as a result of the projected 15 structure deployments at 3 sites over the life of the proposed action as:

**FH 1 (up to 5 deployed structures, 1 reef site):**

Years 26-100:  $75 \text{ years} \times 1 \text{ sea turtle take} = 75 \text{ sea turtle takes per structure} \times 5 \text{ structures} = 375 \text{ turtle mortalities}$

Years 101-150:  $50 \text{ years} \times (1 \text{ sea turtle take} \div 3 \text{ years}) = 16.667 \text{ sea turtle takes} \times 5 \text{ structures} = 83.335 \text{ rounded up for whole organism estimate} = 84 \text{ sea turtle mortalities}$

Total for 150 years:  $375 + 84 = 459 \text{ total sea turtle mortalities}$

**FH 2 (up to 6 deployed structures, 1 reef site):**

Years 26-100:  $75 \text{ years} \times 1 \text{ sea turtle} = 75 \text{ sea turtle takes per structure} \times 6 \text{ structures} = 450 \text{ turtle mortalities}$

Years 101-150:  $50 \text{ years} \times (1 \text{ sea turtle take} \div 3 \text{ years}) = 16.667 \text{ sea turtle takes} \times 6 \text{ structures} = 100.002 \text{ rounded up for whole organism estimate} = 101 \text{ sea turtle mortalities}$

Total for 150 years:  $450 + 101 = 551 \text{ total sea turtle mortalities}$

**FH 7 (up to 4 deployed structures, 1 reef site):**

Years 26-100:  $75 \text{ years} \times 1 \text{ sea turtle} = 75 \text{ sea turtle takes per structure} \times 4 \text{ structures} = 300 \text{ turtle mortalities}$

Years 101-150:  $50 \text{ years} \times (1 \text{ sea turtle take} \div 3 \text{ years}) = 16.667 \text{ sea turtle takes} \times 4 \text{ vessels} = 66.668 \text{ rounded up for whole organism estimate} = 67 \text{ sea turtle mortalities}$

Total for 150 years:  $300 + 67 = 367 \text{ total sea turtle mortalities}$

**Table 8. Entanglement sea turtle mortality estimates for up to 150 years**

<b>High Relief Artificial Reef Site</b>	<b>Sea Turtle Mortality YR 0 to YR 25</b>	<b>Sea Turtle Mortality YR 26 to YR 100</b>	<b>Sea Turtle Mortality YR 101 to YR 150</b>	<b>TOTAL Sea Turtle Mortality YR 0 to YR 150</b>
<b>FH 1 (with 5 structures)</b>	<b>0</b>	<b>375</b>	<b>84</b>	<b>459</b>
<b>FH 2 (with 6 structures)</b>	<b>0</b>	<b>450</b>	<b>101</b>	<b>551</b>
<b>FH 7 (with 4 structures)</b>	<b>0</b>	<b>300</b>	<b>67</b>	<b>367</b>
<b>Total Sea Turtle Mortality by Time Period</b>	<b>0</b>	<b>1,125</b>	<b>252</b>	<b>1,377</b>

In total, the number of sea turtle takes over the course of 150 years as a result of the anticipated number of deployed structures as high relief artificial reef material is estimated to be 1,377 sea turtles.

**5.1.1 Species take percentages**

We used the 2008-2016 STSSN data for the Mississippi Gulf zones 11 and 12 (includes the action area), in order to determine the expected amount of take for each species in the action area. The 9-year dataset for this area shows a total of 2,349 sea turtle strandings (excluding unidentified turtles). Based on the artificial reef location and substrate type, we believe this is the best available data to estimate the relative abundance of sea turtle species in the action area and therefore, the percentages of sea turtle take by species as a result of the proposed action (Table 9).

**Table 9. 2008 – 2016 Mississippi (Gulf) Zones 11 and 12 Sea Turtle Stranding Data**

Species	Total Strandings 2008-2016	Species Percent Composition
Kemp's ridley	2,243	95.5
Loggerhead	72	3.1
Green	33	1.4
Leatherback	1	0.04
Grand Total	2,349	100

To calculate the number of expected sea turtle takes broken down by species, we use the following equation, results of which are summarized in Table 10:

*Expected takes by species for an artificial reef over a 150-year time frame out of 1,377 anticipated sea turtles*

*= total expected sea turtle takes in 150 years from artificial reefs (1,377) x percent composition from stranding data for each species (Table 9)*

*Expected takes for Kemp's ridley sea turtles over 150 years = 1,377 x 0.955 = 1,315.035*

*Expected takes for loggerhead sea turtles over 150 years = 1,377 x 0.031 = 42.687*

*Expected takes for green sea turtles over 150 years = 1,377 x 0.014 = 19.278*

*Expected takes for leatherback sea turtles over 150 years = 1,377 x 0.0004 = 0.5508*

**Table 10. Breakdown of Lethal Sea Turtle Species Entanglement Takes Based on Stranding Data**

Species	Percent from Stranding Data	Species Breakdown Out of 1,377 Anticipated Sea Turtle Takes	Take Estimate Rounded Up
Kemp's ridley	95.5	1315.035	1316
Loggerhead	3.1	42.687	43
Green	1.4	19.278	20
Leatherback	0.04	0.5508	1

*North Atlantic and South Atlantic Green Sea Turtle DPSs*

As described in Section 3.2.1.1, information suggests that the vast majority of the anticipated green sea turtles caught in the Gulf of Mexico and South Atlantic regions are likely to come from the North Atlantic DPS. However, it is possible that animals from the South Atlantic DPS could be captured during the proposed action. We assume based on Foley et al. (2007) that 96% of animals affected by the proposed action are from the North Atlantic DPS and that 4% of the green sea turtles affected by the proposed action are from the South Atlantic DPS. Applying these percentages to our estimated take of 20 green sea turtles over 150 years and rounding in such a way as to conservatively assume the most lethal captures, results in an estimated catch of up to 20 green sea turtles from the North Atlantic DPS (19.278\*0.96=18.507, rounded up to 19), and an estimated catch of up to 1 green sea turtles from the South Atlantic DPS (19.278\*0.04=0.771, rounded up to 1).

Table 11 summarizes the total number of anticipated lethal takes for each species of sea turtle. Using the three time periods for the maturity of reefs and the percentage of species composition, the take for each species during each time period has been calculated. Totals from Table 10 were rounded up from the 10<sup>th</sup> percentile to be most conservative for the species. We note rounding estimates from Table 10 results in a slightly higher total number of sea turtle takes (i.e., 1,380 instead of 1,377).

**Table 11. Anticipated Amount of Lethal Take for Sea Turtle Species due to Entanglements Associated with 15 Structures Deployed As High Relief Artificial Reef Material Over a Period of 150 Years**

Species	Sea Turtle Mortality YR 0 to YR 25	Sea Turtle Mortality YR 26 to YR 100	Sea Turtle Mortality YR 101 to YR 150	TOTAL Sea Turtle Mortality YR 0 to YR 150 <sup>10</sup>
Kemp's ridley sea turtle	0	1074.375	240.66	1316
Loggerhead sea turtle (NWA DPS)	0	34.875	7.812	43
Green sea turtle (NA DPS)	0	15.12	3.386	19
Green sea turtle (SA DPS)	0	0.063	0.141	1
Leatherback sea turtle	0	0.45	0.100	1
<b>Total sea turtle mortality by time period</b>	<b>0</b>	<b>1128</b>	<b>255</b>	<b>1380</b>

## 5.2 Effects to Designated Critical Habitat

Of the artificial reef sites covered under this Opinion, sites FH-8, FH-9, FH-10, and FH-11 are within the boundary of the GSCH Unit 8. Of the 7 possible PCEs that define GSCH, GSCH Unit 8 contains 4, all of which may be affected by the proposed action, namely the placement of low relief artificial reef material of opportunity (i.e., reef balls) in FH-8, FH-9, FH-10 and FH-11. The 4 PCEs that are in GSCH Unit 8 analyzed below are the following: prey abundance (PCE 1); water quality (PCE 2); sediment quality (PCE 3); and migratory pathways (PCE 4).

### *Prey Abundance (PCE 1)*

A number of factors could affect prey abundance (PCE 1) in Gulf sturgeon critical habitat (Peterson et al. 2013). NMFS has determined that the following factors are relevant to the proposed action and hence analyzed in this Opinion: prey items, benthic community structure, and potential Gulf sturgeon in the action area.

### *Prey Items*

Both adult and subadult Gulf sturgeon are known to lose up to 30 percent of their total body weight while in the rivers, and subsequently compensate the loss during winter feeding in estuarine and marine areas (Carr 1983; Clugston et al. 1995; Heise et al. 1999; Morrow et al. 1998; Ross et al. 2000b; Sulak and Clugston 1999; Wooley and Crateau 1985). Therefore, once

<sup>10</sup> The numbers in this column are rounded up to the next whole number.

Gulf sturgeon leave the river after having spent at least six months fasting, it is presumed that they immediately begin feeding. Upon exiting the rivers, Gulf sturgeon concentrate around the mouths of their natal rivers in lakes and bays (depth less than 20 ft). These areas are very important for the Gulf sturgeon as they offer the first foraging opportunity upon exiting the rivers. Gulf sturgeon have been described as opportunistic and indiscriminate benthivores; their guts generally contain benthic marine invertebrates including amphipods, lancelets, polychaetes, gastropods, shrimp, isopods, molluscs, and crustaceans (Carr et al. 1996; Fox et al. 2000; Fox et al. 2002a; Huff 1975; Mason Jr. and Clugston 1993). Generally, Gulf sturgeon prey are burrowing species (e.g., annelids, polychaetes, oligochaetes, amphipods, isopods, and lancelets) that feed on detritus and/or suspended particles, and inhabit sandy substrate. The proposed action will convert approximately 179 ac of existing benthos from muddy sand to concrete, reef balls, and other materials of opportunity and therefore has the potential to affect Gulf sturgeon prey items.

#### *Benthic Community Structure*

According to the USACE Mobile District, the muddy, soft bottom substrates that comprise about 78% of the open water zone of the Mississippi Sound system are inhabited primarily by polychaetes (*Mediomastus ambiseta*, *Steblospio benedicti*) and amphipods (*Grandidierella bonnieroides*). Areas around river mouths typically have much higher numbers of infaunal macroinvertebrates than areas outside of the region of general flow. Such differences have been attributed to deposition of nutrients and detritus by rivers during periods of flooding, and increased activity and abundance of benthic macroinvertebrates. Without a comprehensive benthic survey, current availability of Gulf sturgeon prey within the action area is undeterminable. While the absolute biomass of benthic meio- and macrofauna is not totally dependent upon sediment grain size, community structure and faunal size directly correlate to benthic substrate (Parsons et al. 1984). Because the proposed action will convert up to approximately 179 ac of currently muddy sand within GSCH to concrete and reef balls, the existing benthic community structure will be lost. Therefore, the proposed action has the potential to affect benthic community structure.

#### *Potential Gulf Sturgeon Prey in the Action Area*

Data are available from estuarine/marine habitats throughout the geographic range of Gulf sturgeon that have been assessed for benthic composition. Research in Choctawhatchee Bay indicates that Gulf sturgeon show a preference for sandy shoreline habitats with the majority of fish being located in areas lacking seagrass (Craft et al. 2001; Fox and Hightower 1998; Fox et al. 2002b; Parauka et al. 2001; Rogillio et al. 2007; Ross et al. 2001; Ross et al. 2009). Gulf sturgeon in Pensacola Bay have been found to prefer shallow shoals with unvegetated, fine to medium-grain sand habitats such as sandbars and subtidal energy zones resulting in sediment sorting and a preponderance of sand supporting a variety of prey items (Craft et al. 2001). Habitats used nearby the Mississippi Sound barrier islands tend to have a clean sand substrate and all benthic samples from the area contained lancelets (Craft et al. 2001; Fox and Hightower 1998; Fox et al. 2002b; Parauka et al. 2001; Rogillio et al. 2007; Ross et al. 2001; Ross et al. 2009). Other nearshore Gulf of Mexico locations where Gulf sturgeon are often located consist of unconsolidated, fine-medium grain sand habitats, including natural inlets and passes that are known to support Gulf sturgeon prey items (Abele and Kim 1986; AFS 1989; Menzel 1971). More recent research shows that eastern population Gulf sturgeon were detected overwintering

on shallow, sandy bottom near Mississippi barrier islands (Vick et al. 2018a; Vick et al. 2018b) indicating foraging habitat in this area is used by fish from both eastern and western population segments. Additionally, high prey abundance is commonly associated with shallow (1-3 m), sandy areas from Mississippi Sound eastward (Wilber et al. 2019). Therefore, it has been concluded that Gulf sturgeon are foraging in these sandy areas where they are repeatedly located, as this habitat supports their prey (see preceding section “Prey items” for specifics).

The placement of new artificial reef material onto the existing muddy sand substrate is not expected to reduce the availability of potential prey items in the action area because the existing benthos within the action area is not the preferred forage habitat of Gulf sturgeon (i.e., it is not soft sand). The action area is less functional than adjacent sand areas for predominant Gulf sturgeon prey items, and when the relatively small patchy areas (compared to the entire unit) impacted by the action is modified, the impacts will be insignificant. Therefore, NMFS concludes that the proposed action’s impact on an area of muddy sand benthos is not expected to reduce the critical habitat’s ability to support Gulf sturgeon conservation.

#### *Summary of Effects on Prey Abundance (PCE 1)*

The proposed action may affect the prey abundance PCE in Unit 8 of Gulf Surgeon designated critical habitat. Gulf sturgeon from both the Pearl and Pascagoula rivers are known to forage in Unit 8. The most northwest FH site (FH-8) is approximately 24 mi from the entrance to the Pascagoula River and 37 mi from the entrance to the Pearl River, presumably where Gulf sturgeon forage immediately after exiting the rivers, and 19 miles from the nearest pass, where telemetry data has shown fish from both systems use for winter feeding. The proposed action will convert up to 179 ac of existing benthos from muddy sand to concrete and reef balls. We believe the installation of artificial reef material of opportunity will be insignificant on prey abundance since the area of impact (179 ac) is relatively small compared to the unit as a whole and surrounding area. Prey items will still be present in the surrounding sediment, and while some Gulf sturgeon prey items are found in muddy-sand (e.g., polychaetes), predominant Gulf sturgeon prey items (in terms of frequency and biomass) are classified as burrowing species that inhabit sandy substrate (e.g., lancelets, amphipods, isopods) due to the texture and function of the benthos, and therefore this benthic composition may not be functioning as optimal foraging habitat. Gulf sturgeon are opportunistic suction feeders that tend to forage in calm marine and estuarine waters that support their macroinvertebrate prey including brachiopods, mollusks, worms, and crustaceans (Mason Jr and Clugston 1993), and can easily locate prey and fulfill nutritional requirements in areas adjacent to the action area. During foraging periods Gulf sturgeon generally occupy nearshore areas between 6.5-13 ft (2-4 m) of depth characterized by low-relief sand substrate (Fox et al. 2002a), which is are mostly shallower than depths within the deployment area of the proposed action that range from 12 to 16 ft (~4 to 5 m).

#### *Water Quality (PCE 2)*

NMFS believes effects to the water quality PCE in Unit 8 of GSCH as a result of new artificial reef material placement will be insignificant. Impacts from sediment disturbance during artificial reef deployment are expected to be temporary and minimal, with suspended particles settling out within a short time frame and no measurable effects on water quality. No changes in temperature, salinity, pH, hardness, oxygen content, and other chemical characteristics are expected from this proposed action.

### *Sediment Quality (PCE 3)*

Potential effects to sediment quality resulting from sediment modification by this project can permanently impact the availability and abundance of prey. During foraging periods, Gulf sturgeon generally occupy nearshore areas between 6.5-13 ft (2-4 m) of depth characterized by low-relief substrate (Fox et al. 2002a). Although there will be adverse effects to the sediment quality from the deployment of artificial reef material, the project area does not seem to be functioning as preferred habitat for sturgeon as discussed above, and sturgeon will be able to access adjacent habitats with similar sediment quality. Thus, we believe the placement of low relief artificial reef material of opportunity (i.e., reef balls) in FH-8, FH-9, FH-10 and FH-11 will adversely affect 179 ac of Unit 8 of Gulf sturgeon designated critical habitat by permanently converting the sediment quality from silty and muddy sand to concrete, reef balls and other materials of opportunity.

### *Migratory Pathways (PCE 4)*

Unit 8 is known to support migratory pathways for Gulf sturgeon for 2 western subpopulations (Pascagoula and Pearl rivers), as groups of individuals from these subpopulations have been located by acoustic telemetry on numerous occasions (Brooks and Sulak 2005; Fox et al. 2002b; Ross et al. 2000a; Sulak et al. 2009; Vick et al. 2018a); Harris, Parkyn, and Murie, 2005. Subadult and adult Gulf sturgeon move from spawning rivers in the spring through estuarine and marine areas to feeding areas in the winter. There is also potential for inter-riverine movement between these rivers. Gulf Sturgeon from the eastern population segment overwinter in habitats that are characterized as nearshore shallow areas (2 to 4 m) with sandy bottoms (Fox et al. 2002b), which is prevalent at the Mississippi barrier islands (Ross et al. 2009). In Ross et al. (2009), only western population Gulf Sturgeon were detected using shallow, sandy bottom areas surrounding the Mississippi barrier islands; no Gulf sturgeon were detected in the middle of the Mississippi Sound or at inshore, non-island sites. In contrast, Peterson et al. (2013) documented high occupancy nearshore in the western distributary of the Pascagoula River estuary; these habitats had finer grain size characteristics compared to Mississippi barrier islands and eastern population segment habitats (Brooks and Sulak 2005; Fox et al. 2002b; Ross et al. 2000a; Sulak et al. 2009) Harris, Parkyn, and Murie, 2005. Therefore, habitat type and sediment composition in nearshore areas may be different between eastern and western populations of Gulf sturgeon (Peterson et al. 2013). Gulf sturgeon are characterized as benthic cruisers and it is presumed that the fish are more concentrated in available sandy areas due to a high accessibility of prey items (Fox et al. 2002b; Parauka et al. 2011; Peterson et al. 2013; Ross et al. 2000a). Additionally, overwintering adult and subadult Gulf sturgeon from the Pearl, Pascagoula, Escambia, Blackwater, Yellow, and Choctawhatchee rivers have been found to occupy the habitat around Ship Island for foraging (Vick et al. 2018b).



The artificial reef sites within the boundary of the GSCH Unit 8 included in the proposed action will convert approximately 179 ac of submerged benthos to hardbottom. In this Opinion, we assume that the entire 179 ac will be converted. New reef material may block passage within and between riverine, estuarine, and marine habitats; however, NMFS believes effects to the migratory pathway PCE in Unit 8 of GSCH as a result of the proposed action will be insignificant. The patchy sites within the action area are distanced far enough from the nearest river mouth (approximately 24 mi from the entrance to the Pascagoula River and 37 mi from the entrance to the Pearl River) where sturgeon will be able to swim around the sites and continue to have access to riverine, estuarine, and marine habitats.

### **5.3 Summary of the Effects of the Action on GSCH PCEs**

Combining the total impacts to Unit 8 of Gulf sturgeon designated critical habitat from the loss of the sediment quality essential feature, we believe the proposed action will adversely affect approximately 179 ac of Gulf sturgeon critical habitat, which is approximately 0.02 percent of the total area within Unit 8. NMFS analyzed the proposed action's effects on GSCH PCEs, and concluded that the project activity would have insignificant effects on the prey abundance, water quality, and migratory pathways PCEs. NMFS also concluded that the deployment of artificial reef material would have adverse effects on the sediment quality PCE. While the sediment quality at the artificial reef sites FH-8, FH-9, FH-10 and FH-11 will be diminished, these sites do not seem to be functioning as preferred habitat for Gulf sturgeon and the proposed action is not preventing Gulf sturgeon from accessing alternate habitat with the sediment quality essential feature in the surrounding area. Still some ecological function provided to Gulf sturgeon in terms of the sediment quality will be lost; therefore, we believe the project is likely to adversely affect 179 ac of Gulf sturgeon designated critical habitat in Unit 8.

## **6 CUMULATIVE EFFECTS**

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Cumulative effects include the effects of *future* state, tribal, or local private actions that are reasonably certain to occur in the action area considered in this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA (50 CFR 402.14). Actions that are reasonably certain to occur would include actions that have some demonstrable commitment to their implementation, such as funding, contracts, agreements or plans.

### **Cumulative Effects to ESA-Listed Sea Turtles**

Human-induced mortality and/or injury of sea turtles occurring in the action area are reasonably certain to occur in the future. Primary sources of those effects include vessel interactions and pollution. While the combination of these activities may prevent or slow the recovery of populations of sea turtles, the magnitude of these effects is currently unknown.

#### *Vessel Interactions*

NMFS's STSSN data indicate that vessel interactions are responsible for a large number of sea turtles stranding within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many

stranded sea turtles have obvious propeller or collision marks (Dwyer et al. 2003). Still, it is not always clear whether the collision occurred pre- or post-mortem. We believe that sea turtle injuries and mortalities by vessel interactions will continue in the future.

### Pollution

Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from the pollution on sea turtles. However, the level of impacts cannot be projected. Marine debris (e.g., discarded fishing line or lines from boats) can entangle sea turtles in the water and drown them. Sea turtles commonly ingest plastic or mistake debris for food.

Noise pollution has been raised primarily as a concern for marine mammals (including ESA-listed large whales) but may be a concern for other marine organisms, including sea turtles. The potential effects of noise pollution on sea turtles range from minor behavioral disturbance to injury and death. The noise level in the ocean is thought to be increasing at a substantial rate due to increases in shipping and other activities, including seismic exploration, offshore drilling, and sonar used by military and research vessels. Concerns about noise in the action area of this consultation include increasing noise due to increasing recreational vessels.

Beyond the threats noted above, NMFS is not aware of any proposed or anticipated changes in other human-related actions (e.g., poaching, habitat degradation) or natural conditions (e.g., overabundance of land or sea predators, changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on the sea turtles covered by this Opinion.

### **Cumulative Effects to Gulf Sturgeon Designated Critical Habitat**

As noted in Section 4 discussing the Environmental Baseline, effects on Gulf sturgeon designated critical habitat related to the DWH oil spill are not fully known at this time. Routes of exposure are generally believed to be, 1) suffocation of infaunal organisms, and 2) toxicity of substrate. Both of these effects would impact potential foraging areas within Gulf sturgeon critical habitat through the displacement and/or reduction of prey items. Some emergency restoration efforts pursuant to the Oil Pollution Act have yet to be determined and implemented, and so the ultimate restoration impacts on the species are unknowable at this time.

Coastal runoff and river discharges carry large volumes of petrochemical and other contaminants from agricultural activities, cities, and industries into the Gulf of Mexico. The coastal waters of the Gulf of Mexico have more sites with high contaminant concentrations than other areas of the coastal United States due to the large number of waste discharge point sources. Chemicals and metals such as chlordane, dichlorodiphenyldichloroethylene, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the substrate and are later incorporated into the food web as they are consumed by benthic feeders, such as sturgeon or macroinvertebrates. Some of these compounds may affect the surrounding environment by reducing DO, altering pH, and altering other water quality properties.

Although little is known about contaminant effects on Gulf sturgeon critical habitat general studies on sturgeon habitats indicate that the effects of contaminants and pollution contribute to lost habitat (Barannikova 1995; Shagaeva et al. 1993; Verina and Peseridi 1979).

## 7 INTEGRATION AND SYNTHESIS JEOPARDY ANALYSIS AND DESTRUCTION/ADVERSE MODIFICATION ANALYSIS

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### 7.1 Jeopardy Analysis

The analyses conducted in the previous sections of this Opinion provide the basis on which we determine whether the proposed action would likely jeopardize the continued existence of ESA-listed sea turtle species. In the Effects of the Action section, we outlined how the proposed action, specifically the presence of high-relief artificial reef material of opportunity at 3 of the FH sites, would affect these species at the individual level and the magnitude of those effects based on the best available data. Next, we assessed each of these species' response to the effects of the proposed action in terms of overall population effects and whether those effects will jeopardize their continued existence in the context of the status of the species, the environmental baseline, and the cumulative effects.

It is the responsibility of the action 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 species or threatened species...” (ESA Section 7(a)(2)). Action agencies must consult with and seek assistance from the NMFS to meet this responsibility. NMFS must ultimately determine in a Biological Opinion whether the action jeopardizes listed species. To *jeopardize the continued existence of* is defined as “to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species” (50 CFR 402.02). The following jeopardy analysis first considers the effects of the action to determine if we would reasonably expect the action to result in reductions in reproduction, numbers, or distribution of ESA-listed sea turtles. The analysis next considers whether any such reduction would in turn result in an appreciable reduction in the likelihood of survival of these species in the wild, and the likelihood of recovery of these species in the wild.

#### 7.1.1 Kemp's Ridley Sea Turtles

The proposed action may result in the lethal take of 1,316 Kemp's ridley sea turtles over the next 150 years. The take is expected to be no Kemp's ridley sea turtles during the first 25 years, approximately 1,074 during the next 75 years (one turtle per year per structure deployed), and approximately 240 during the last 50 years (one turtle every 3 years per structure deployed).

##### 7.1.1.1 Survival

The potential lethal take of up to 1,316 Kemp's ridley sea turtles over the next 150 years as a result of the proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. The TEWG (Turtle Expert Working Group 1998b) estimates age at maturity from 7-15 years, females return to their nesting beach about every 2 years (Turtle Expert Working Group 1998b). The mean clutch size for Kemp's ridley sea turtle is 100 eggs/nest, with an average of 2.5 nests/female/season. As a result, lethal takes could also result in a potential reduction in future reproduction, assuming at least some of the individuals lethally taken are female and would have otherwise survived to reproduce in the future. The loss of 1,316 Kemp's

ridley sea turtles could preclude the production of thousands of eggs and hatchlings, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their contribution to future generations, and result in a reduction in sea turtle reproduction. The anticipated lethal takes are expected to occur over a long time period (150 years), with more than 80% of those takes occurring after the artificial reef sites become mature (25 years) and before the artificial reef sites reach the age of 100. In addition, the deployment of high-relief artificial reef material will occur in a discrete area (i.e., FH-1, FH-2 and FH-7). Because Kemp's ridley sea turtles generally have large ranges, no reduction in the distribution is expected from the take of these individuals over the life of the proposed action.

In the absence of any total population estimates for Kemp's ridley sea turtle, nesting trends are the best proxy for estimating population changes. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley sea turtle nests in Mexico reached a record high of 21,797 in 2012 (Gladys Porter Zoo nesting database 2013). There was a second significant decline in Mexico nests 2013 through 2014; however, nesting in Mexico has increased 2015 through 2017 (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, followed by another decline to 11,090 in 2019 (Gladys Porter Zoo 2019).

A small nesting population is also emerging in the United States, primarily in Texas, rising from 4 nests in 1995 to 197 in 2009, to a record high of 353 nests in 2017 [(NMFS and USFWS 2015); (NPS 2017)]. Nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015-2017, and then a drop back down to 190 nests in 2019.

Given the significant inter-annual variation in nesting data, sea turtle population trends necessarily are measured over decades and the long-term trend line better reflects the population increase in Kemp's ridleys. With the recent increase in nesting data (2015-17) and recent declining numbers of nesting females (2013-14 and 2018-19), it is too early to tell whether the long-term trend line is affected. Nonetheless, long-term data from 1990 to present continue to support that Kemp's ridley sea turtle is increasing in population size.

We believe this long-term increasing trend in nesting is evidence of an increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information is clearly increasing, we believe the potential lethal take of 1,316 Kemp's ridley sea turtles over the next 150 years attributed to the proposed action 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 is not reasonably expected to cause an appreciable reduction in the likelihood of survival of Kemp's ridley sea turtles in the wild.

### 7.1.1.2 Recovery

As to whether the proposed action 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:

*Objective: A population of at least 10,000 nesting females in a season (as measured by clutch frequency/female/season) distributed at the primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) in Mexico is attained. Methodology and capacity to implement and ensure accurate nesting female counts have been developed.*

With respect to this recovery objective, the most recent nesting numbers in 2019 indicate there were a total of 11,090 nests on the main nesting beaches in Mexico. This number represents approximately 4,436 nesting females for the season based on 2.5 clutches/female/season. The number of nests reported annually from 2010 to 2014 overall declined, rebounded in 2015 through 2017, but have 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 up to 1,316 Kemp's ridley sea turtles over the next 150 years (with no takes anticipated during the first 25 years) is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede the progress toward achieving this recovery objective. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of Kemp's ridley sea turtles' recovery in the wild.

### 7.1.1.3 Conclusion

The lethal take of 1,316 Kemp's ridley sea turtles associated with the proposed action over the next 150 years (with no takes anticipated during the first 25 years) is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of Kemp's ridley sea turtle in the wild.

## 7.1.2 Loggerhead Sea Turtles (NWA DPS)

The proposed action may result in the lethal take of 43 loggerhead sea turtles from the NWA DPS over the next 150 years. The take is expected to be no loggerhead sea turtles during the first 25 years, approximately 35 takes during the next 75 years (less than one turtle per year), and approximately 8 takes during the last 50 years (less than one turtle every 3 years).

### 7.1.2.1 Survival

The potential lethal take of up to 43 loggerhead sea turtles over the next 150 years as a result of the proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. A lethal take could also result in a potential reduction in future reproduction, assuming at least some of the individuals taken are 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 years, with 100-126 eggs per clutch. While we have no reason to believe the proposed action will disproportionately affect females, the loss of even 1 adult female could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. The anticipated lethal takes are expected to occur over a long time period (150 years), 35 those takes occurring after the artificial reef sites become mature (25 years) and before the artificial reef sites reach the age of 100. Therefore, a reduction in the distribution of loggerhead sea turtles is not expected from lethal takes attributed to the proposed action. In addition, the anticipated lethal take is expected to occur in a discrete area and loggerhead sea turtles in the NWA DPS generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether or not the reductions in loggerhead sea turtle numbers and reproduction attributed to the proposed action would appreciably reduce the likelihood of survival depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends (i.e., whether the estimated reductions, when viewed within the context of the environmental baseline, the status of the species and cumulative effects, are of such an extent that adverse effects on population dynamics are appreciable). In Section 3.2.1.3, we reviewed the status of this species in terms of nesting and female population trends and several recent assessments based on population modeling (i.e., (Conant et al. 2009; NMFS 2009). Below we synthesize what that information means both in general terms and the more specific context of the proposed action.

Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerhead sea turtles require high survival rates throughout their life to maintain a population. In other words, late-maturing species cannot tolerate much anthropogenic mortality without going into decline. Conant et al. (2009) concluded 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 is slow, 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; Heppell et al. 1995).

NMFS (2009) estimated the minimum adult female population size for the NWA DPS in the 2004-2008 timeframe to likely be between approximately 20,000-40,000 individuals (median 30,050), with a low likelihood of being as many as 70,000 individuals. Another estimate for the entire western North Atlantic population was a mean of 38,334 adult females using data from 2001-2010 (Richards et al. 2011). A much less robust estimate for total benthic females in the western North Atlantic was also obtained, with a likely range of approximately 30,000-300,000 individuals, up to less than 1 million.

NMFS (2011) preliminarily estimated the loggerhead population in the Northwestern Atlantic Ocean along the continental shelf of the Eastern Seaboard during the summer of 2010 at 588,439 individuals (estimate ranged from 381,941 to 817,023) based on positively identified individuals. The NMFS-NEFSC's point estimate increased to approximately 801,000 individuals when including data on unidentified sea turtles that were likely loggerheads. The NMFS-NEFSC (2011) underestimates the total population of loggerheads since it did not include Florida's east coast south of Cape Canaveral or the Gulf of Mexico, which are areas where large numbers of

loggerheads are also expected. In other words, it provides an estimate of a subset of the entire population.

Florida accounts for more than 90% of U.S. loggerhead nesting. The FWC conducted a detailed analysis of Florida's long-term loggerhead nesting data (1989-2019). They indicated that following a 24% increase in nesting between 1989 and 1998, nest counts declined sharply from 1999 to 2007. However, annual nest counts showed a strong increase (71%) from 2008 to 2016. Examining only the period between the high-count nesting season in 1998 and the 2016 nesting season, researchers found a slight but insignificant increase, indicating a reversal of the post-1998 decline. Nesting in 2017 declined relative to 2016, back to levels seen in 2013 and 2015, but rose slightly in 2018 and 2019. The overall change in counts from 1989 to 2019 was significantly positive; however, it should be noted that wide confidence intervals are associated with this complex data set, which, along with uncertainty around the variability in nesting parameters (nests/female, nesting intervals, etc.) it is unclear whether the positive nesting trend equates to an increase in the population of nesting females over that time frame (Ceriani et al. 2019).

Abundance estimates accounting for only a subset of the entire loggerhead sea turtle population in the western North Atlantic indicate the population is large (i.e., several hundred thousand individuals). Nesting trends have been significantly increasing over several years against the background of the past and ongoing human and natural factors (as contemplated in the Status of the Species and Environmental Baseline) that have contributed to the current status of the species.

The proposed action could lethally take 43 loggerhead sea turtles over the next 150 years. We do not expect this loss to result in a detectable change to the population numbers or increasing trends because this loss is anticipated to occur over a long timeframe and would result in a low amount of take on an average annual basis compared to the total population estimate and anticipated growth rate. . 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 is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the loggerhead sea turtle NWA DPS in the wild.

#### *7.1.2.2 Recovery*

The loggerhead recovery plan for the Northwest Atlantic population of loggerhead sea turtles defines the recovery goal as "...ensur[ing] that each recovery unit meets its Recovery Criteria alleviating threats to the species so that protection under the ESA is no longer necessary" (NMFS and USFWS 2008). The plan then identifies 13 recovery objectives needed to achieve that goal.

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2008) lists the following recovery objectives that are relevant to the effects of the proposed action:

*Objective: Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females.*

*Objective: 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 western North Atlantic population will recover within 50-150 years, but notes that reaching recovery in only 50 years would require a rapid reversal of the then-declining trends of the NRU, PFRU, and NGMRU. The minimum end of the range assumes a rapid reversal of the current declining trends; the higher end assumes that additional time will be needed for recovery actions to bring about population growth (NMFS and USFWS 2008).

Nesting trends in most recovery units have been significantly increasing over several years. We do not believe the proposed action impedes the progress of the recovery program or achieving the overall recovery strategy because the amount of take expected to occur over a 150-year time period, as a result of the proposed action is not expected to be detectable on a population level or on nesting trends, and therefore it is not expected to affect population growth over the timeframe analyzed. We also indicated that the lethal take of 43 loggerhead sea turtles over the next 150 years is so small in relation to the overall population, that it would not impede achieving the Recovery Objectives, even when considered in the context of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. We believe this is true for both nesting and juvenile in-water populations. For these reasons, we do not believe the proposed action will impede achieving the recovery objectives or overall recovery strategy.

#### *7.1.2.3 Conclusion*

The lethal take of 43 loggerhead sea turtles associated with the proposed action over the next 150 years (with no takes anticipated during the first 25 years) is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NWA DPS of the loggerhead sea turtle in the wild.

### **7.1.3 Green Sea Turtles**

The proposed action may result in the lethal take of 20 green sea turtles (19 from the NA DPS and 1 from the SA DPS) over the next 150 years. The take is expected to be no green sea turtles during the first 25 years, 16 during the next 75 years, and 4 during the last 50 years.

As discussed in the Effects of the Action section, green sea turtles from both the NA and SA DPSs can be found on foraging grounds within U.S. waters. While there are currently no in-depth studies available to determine the percent of NA and SA DPS individuals in any given location, an analysis of cold-stunned green turtles off the St. Joseph Bay, Florida, foraging grounds, which is located on the northern coast of the Gulf of Mexico, found approximately 4% of juvenile individuals came from nesting stocks in the SA DPS (specifically Suriname/Aves Island, Brazil, Ascension Island, and Guinea Bissau) (Foley et al. 2007). While it is highly likely green sea turtles found in or near the action area will be from the NA DPS, we cannot rule out that they may also be from the SA DPS. Therefore, to analyze effects in a precautionary manner, we will conduct 2 jeopardy analyses, one for each DPS (i.e., assuming up to 96% could come from the NA DPS and 4% could come from the SA DPS).



### 7.1.3.1 NA DPS

The proposed action may result in the lethal take of 19 green sea turtles from the NA DPS over the next 150 years.

#### 7.1.3.1.1 Survival

The potential lethal take of up to 19 green sea turtles from the NA DPS over the next 150 years as a result of the proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. A lethal take could also result in a potential reduction in future reproduction, assuming that at least some of the individuals take are female and would have survived to reproduce in the future. For example, as discussed above, 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 is expected to survive to sexual maturity. The anticipated lethal takes are expected to occur over a long time period (150 years) with more than 18 of those takes occurring after the artificial reef sites become mature (25 years) and before the artificial reef sites reach the age of 100. In addition, the deployment of the high-relief artificial reef material will occur in a discrete area (i.e., FH-1, FH-2 and FH-7). Because green sea turtles from the NA DPS generally have large ranges, 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.

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, USA (approximately 8,400 nesters; 5%), also accounting for a large portion of the overall nesting (Seminoff et al. 2015). At Tortuguero, Costa Rica, the number of nests laid per year from 1999 to 2010 increased, despite substantial human impacts to the population at the nesting beach and at foraging areas (Campbell and Lagueux 2005; Troëng 1998; Troëng and Rankin 2005). Nesting locations in Mexico along the Yucatan Peninsula also indicate the number of nests laid each year increased to over 1,500 nests/year by 2000 (NMFS and USFWS 2007a). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpubl. data, 2013, in Seminoff et al. 2015). In Florida, most nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, FFWCC, pers. comm., 2013). As described in the Section 3.2.1.2, nesting has increased substantially over the last 20 years and peaked in 2019 with almost 41,000 nests on the Index Nesting Beaches (<https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/>).

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. Therefore, we believe the potential lethal take of 19 green sea turtles from the NA DPS over the next 150 years will not have any measurable effect on that trend because this loss is anticipated to occur over a long timeframe and would result in a low amount of take on an average annual basis compared to the increasing 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 is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle NA DPS in the wild.

#### 7.1.3.1.2 Recovery

The NA DPS of green sea turtles does not have a separate recovery plan at this time. However, an Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) does exist. Since the animals within the NA DPS all occur in the Atlantic Ocean and would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide until a new plan, specific to the NA DPS, is developed. The Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

*Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.*

*Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.*

According to data collected from Florida's index nesting beach survey from 1989-2019, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of almost 41,000 in 2019, indicating that the first listed recovery objective is currently being met. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting, however, it is likely that numbers on foraging grounds have also increased, consistent with the criteria of the second listed recovery objective.

The potential lethal take of up to 19 green sea turtles from the NA DPS over the next 150 years as a result of the proposed action will result in a reduction in numbers when it occurs. This take is unlikely to have any detectable influence on the recovery objectives and trends noted above, and will not result in an appreciable reduction in the likelihood of NA DPS green sea turtles' recovery in the wild even when considered in the context of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion.

#### 7.1.3.1.3 Conclusion

The lethal take of 19 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 turtle in the wild.

### 7.1.3.2 SA DPS

The proposed action may result in the lethal take of 1 green sea turtle from the SA DPS over the 150 years as a result of the artificial reefs.

#### 7.1.3.2.1 Survival

The potential lethal take of up to 1 green sea turtle from the SA DPS over the next 150 years as a result of the proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. As discussed above, lethal interactions would also result in a potential reduction in future reproduction, assuming the individual taken is female and would have survived otherwise to reproduce. The anticipated lethal take is expected to occur over a long time period (150 years). In addition the deployment of the high-relief artificial reef material will occur in a discrete area (i.e., FH-1, FH-2 and FH-7). Because green sea turtles in the SA DPS generally have large ranges, no reduction in their 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 3.2.1.2, we summarized available information on number of nesters and nesting trends at SA DPS beaches. Seminoff et al. (2015) estimated that there are greater than 63,000 nesting females in the SA DPS, though they noted the adult female nesting abundance from 37 beaches could not be quantified. The nesting at Poilão, Guinea-Bissau, accounted for approximately 46% of that estimate (approximately 30,000 nesters), with Ascension Island, United Kingdom, (approximately 13,400 nesters; 21%), and the Galibi Reserve, Suriname (approximately 9,400 nesters; 15%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015).

Seminoff et al. (2015) reported that while trends cannot be estimated for many nesting populations due to the lack of data, they could discuss possible trends at some of the primary nesting sites. Seminoff et al. (2015) indicated that the nesting concentration at Ascension Island (United Kingdom) is one of the largest in the SA DPS and the population has increased substantially over the last 3 decades (Broderick et al. 2006; Glen et al. 2006). Mortimer and Carr (1987) counted 5,257 nests in 1977 (about 1,500 females), and 10,764 nests in 1978 (about 3,000 females) whereas from 1999–2004, a total of about 3,500 females nested each year (Broderick et al. 2006). Since 1977, numbers of nests on 1 of the 2 major nesting beaches, Long Beach, have increased exponentially from around 1,000 to almost 10,000 (Seminoff et al. 2015). From 2010 to 2012, an average of 23,000 nests per year was laid on Ascension (Seminoff et al. 2015). Seminoff et al. (2015), caution that while these data are suggestive of an increase, historic data from additional years are needed to fully substantiate this possibility.

Seminoff et al. (2015) reported that the nesting concentration at Galibi Reserve and Matapica in Suriname was stable from the 1970s through the 1980s. From 1975–1979, 1,657 females were counted (Schulz 1982), a number that increased to a mean of 1,740 females from 1983–1987

(Ogren 1989b), and to 1,803 females in 1995 (Weijerman et al. 1996). Since 2000, there appears to be a rapid increase in nest numbers (Seminoff et al. 2015).

In the Bijagos Archipelago (Poilão, Guinea-Bissau), Parris and Agardy (1993 as cited in Fretey 2001) reported approximately 2,000 nesting females per season from 1990 to 1992, and Catry et al. (2002) reported approximately 2,500 females nesting during the 2000 season. Given the typical large annual variability in green sea turtle nesting, Catry et al. (2009) suggested it was premature to consider there to be a positive trend in Poilão nesting, though others have made such a conclusion (Broderick et al. 2006). Despite the seeming increase in nesting, interviews along the coastal areas of Guinea-Bissau generally resulted in the view that sea turtles overall have decreased noticeably in numbers over the past two decades (Catry et al. 2009). In 2011, a record estimated 50,000 green sea turtle clutches were laid throughout the Bijagos Archipelago (Seminoff et al. 2015).

In summary, nesting at some of the primary nesting beaches for the SA DPS has been increasing over the past 3 decades, against the background of the past and ongoing human and natural factors (as contemplated in the Status of the Species and Environmental Baseline sections) 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 green sea turtles is increasing, we believe the potential lethal take of 1 green sea turtle from the SA DPS over the next 150 years as a result of the project 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 is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle SA DPS in the wild.

#### 7.1.3.2.2 Recovery

Like the NA DPS, the SA DPS of green sea turtles does not have a separate recovery plan in place at this time. However, an Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) does exist. Since the animals within the SA DPS all occur in the Atlantic Ocean and would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide until a new plan, specific to the SA DPS, is developed. In our analysis for the NA DPS, we stated that the Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

*Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.*

*Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.*

The nesting recovery objective is specific to the NA DPS, but demonstrates the importance of increases in nesting to recovery. As previously stated, nesting at some of 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 increases in nesting and in-water abundance, however, it is likely that numbers on foraging grounds have increased.

The potential lethal take of 1 green sea turtle from the SA DPS over the next 150 years as a result of the proposed action will result in a reduction in numbers when it occurs, 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. Thus, the proposed action will not impede achieving 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.

#### 7.1.3.2.3 Conclusion

The potential lethal take of 1 SA DPS green sea turtle associated with the proposed action is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of green sea turtle in the wild.

### 7.1.4 **Leatherback Sea Turtles**

The proposed action may result in up to 1 leatherback sea turtle lethal take over the next 150 years. The take is expected to be no leatherback sea turtles during the first 25 years, 1 during the next 75 years, and none during the last 50 years.

#### 7.1.4.1 *Survival*

The lethal take of up to 1 leatherback sea turtle over the next 150 years as a result of the project would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Lethal captures could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and would have survived otherwise to reproduce in the future. For example, an adult female leatherback sea turtle can produce up to 700 eggs or more per nesting season (Schulz 1975). Although a significant portion (up to approximately 30%) of the eggs can be infertile, the annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. While we have no reason to believe the proposed action will disproportionately affect females, the death of any female leatherbacks that would have survived otherwise to reproduce would eliminate its and its future offspring's contribution to future generations. The anticipated lethal take is expected to occur over a long time period (150 years). In addition, the deployment of the high-relief artificial reef material will occur in a discrete area (i.e., FH-1, FH-2 and FH-7). Because leatherback sea turtles generally have large ranges, no reduction in the distribution is expected from the take of these individuals.

The Leatherback TEWG estimated there are between 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic based on 2004 and 2005 nesting count data (Turtle Expert Working Group 2007). The potential loss of up to 1 leatherback sea turtle over the next 150 years accounts for only 0.00001-0.00003% of those population

estimates, which are only a subset of the entire population. We do not believe this potential loss will have any detectable impact on these population numbers.

Of the 15 leatherback nesting populations in the North Atlantic, 7 show an increase in nesting (Florida, Puerto Rico [not Culebra], St. Croix-U.S. Virgin Islands, British Virgin Islands, Trinidad, Guyana, and Brazil) and 3 have shown a decline in nesting (Puerto Rico [Culebra], Costa Rica [Tortuguero], and Costa Rica [Gandoca]). However, subsequent analysis using data up through 2017 has shown decreases in this stock, with an annual geometric mean decline of 10.43% over what they described as the short term (2008-2017) and a long-term (1990-2017) annual geometric mean decline of 5% (NWALWG 2018).

The main nesting areas in Puerto Rico are at Fajardo on the main island of Puerto Rico and on the island of Culebra. Between 1978 and 2005, nesting increased in Puerto Rico from a minimum of 9 nests recorded in 1978 and to a minimum of 469-882 nests recorded each year between 2000 and 2005 (NMFS and USFWS 2013b). However since 2004, nesting has steadily declined in Culebra, which appears to reflect a shift in nest site fidelity rather than a decline in the female population (NMFS and USFWS 2013b).

In the U.S. Virgin Islands, St. Croix (Sandy Point NWR), leatherback nesting was estimated to increase at 13% per year from 1994 through 2001. However, nesting data from 2001 through 2010 indicate nesting has slowed, possibly due to fewer new recruits and lowered reproductive output (NMFS and USFWS 2013b). The average annual growth rate was calculated as approximately 1.1 (with an estimated confidence interval between 1.07 and 1.13) using the number of observed females at Sandy Point, St. Croix, from 1986 to 2004 (Turtle Expert Working Group 2007).

In Costa Rica, Tortuguero, leatherback nesting has decreased 88.5% overall from 1995 through 2011 (NMFS and USFWS 2013b). Troëng et al. (2007) estimated a 67.8% overall decline from 1995 through 2006. However, these estimates are based on an extrapolation of track survey data, which has consistently underestimated the number of nests reported during the surveys (NMFS and USFWS 2013b). Regardless of the method used to derive the estimate, the number of nests observed over the last 17 years has declined. Troëng et al. (2005) found a slight decline in the number of nests at Gandoca, Costa Rica, between 1995 and 2003, but the confidence intervals were large. Data between 1990 and 2004 at Gandoca averaged 582.9 (+ 303.3) nests each year, indicating nest numbers have been lower since 2000 (Chacón-Chaverri and Eckert 2007), and the numbers are not increasing (Turtle Expert Working Group 2007).

Aside from the long-term nesting trend in Florida (an annual geometric mean increase of over 9%), most all of the other nesting populations appear to be decreasing, reversing the stable and increasing trend that was observed as of 2017. However, since we anticipate only 1 mortality over the next 150 years, which is only a tiny fraction of the reduced but still large overall nesting population, and we have no reason to believe nesting females will be disproportionately affected, we believe the potential mortality associated with the proposed action will have no detectable effect on current nesting trends.

Since we do not anticipate the proposed action will have any detectable impact on the population overall, or current nesting trends, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

#### *7.1.4.2 Recovery*

The Atlantic recovery plan for the U.S. population of the leatherback sea turtles (NMFS and USFWS 1992) lists the following relevant recovery objective:

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

We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. As discussed in 3.2.1.5, updated data from 2018 has shown a reverse in trends, as the Culebra, St. Croix, and Florida nesting populations have decreased in recent years; however, it is unclear whether declines may at least in part reflect a shift in nest site fidelity or if it is indicative of a decline in the female population. Broader nesting declines elsewhere on the NW Atlantic nesting beaches suggest that the declines in nests may indicate a true decline in either nesters or reproductive output. However, since we concluded that the potential loss of up to 1 leatherback sea turtle over the next 150 years is not likely to have any detectable effect on these nesting trends, we do not believe the proposed action would impede the progress toward achieving this recovery objective. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild.

#### *7.1.4.3 Conclusion*

The potential lethal take of 1 leatherback sea turtle associated with the proposed action is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the leatherback sea turtle in the wild.

## **7.2 Critical Habitat Destruction/Adverse Modification Analysis**

NMFS's regulations define *Destruction or adverse modification* to mean "a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species" (50 CFR 402.02). Alterations that may destroy or adversely modify critical habitat may include impacts to the area itself, such as those that would impede access to or use of the essential features. NMFS will generally conclude that a Federal action is likely to "destroy or adversely modify" designated critical habitat if the action results in an alteration of the quantity or quality of the essential physical or biological features of designated critical habitat and if the effect of the alteration is to appreciably diminish the value of critical habitat for the conservation of the species.

This analysis takes into account the geographic and temporal scope of the proposed action, recognizing that "functionality" of critical habitat necessarily means that it must now and must continue in the future to support the conservation of the species and progress toward recovery. The analysis takes into account any changes in amount, distribution, or characteristics of the

critical habitat that will be required over time to support the successful recovery of the species. Destruction or adverse modification does not depend strictly on the size or proportion of the area adversely affected, but rather on the role the action area and the affected critical habitat serves with regard to the function of the overall critical habitat designation, and how that role is affected by the action.

The following rationale demonstrates that the proposed action will not impede ability of Unit 8 of Gulf sturgeon critical habitat to support Gulf sturgeon conservation, despite permanent adverse effects to the sediment quality PCE of Gulf sturgeon designated critical habitat. Unit 8 is expected to continue to serve its intended conservation role for Gulf sturgeon.

The placement of low-relief artificial reef material associated with 4 of the FH sites (FH-8, FH-9, FH-10, and FH-11) will impact up to 179 ac of unvegetated substrate that supports prey items (benthic invertebrates) of the prey abundance PCE, the water quality PCE, sediment quality PCE, and the migratory pathways PCE. NMFS concluded that the proposed action would have insignificant effects on the prey abundance, water quality, and migratory pathways PCEs. While Gulf sturgeon may use the affected area, the 179 ac may not be functioning in a way that the species would need for survival and growth. These benthic areas within the project sites are not likely to be regularly used by Gulf sturgeon for feeding due to the substrate characteristics (muddy sand rather than just soft sand), and due to the opportunistic feeding behavior of Gulf sturgeon, this species will be able to utilize more optimal and desired substrate nearby.

Therefore, NMFS concludes that the proposed action's impact on sediment quality is not expected to reduce the critical habitat's ability to support Gulf sturgeon conservation. The conservation function of Unit 8 will remain intact.

Finally, the proposed action will not interfere with recovery objectives, actions, or tasks identified in the Gulf sturgeon recovery plan (USFWS and GSMFC 1995). The proposed action will not affect population size or distribution, disrupt research activities, and will not impede recovery of the species. NMFS concludes that the effects of the project will not discernibly impact the ecological function of Unit 8, and that it will continue to serve its intended conservation role for Gulf sturgeon.

## **8 CONCLUSION**

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Using the best available data, we analyzed the Effects of the Action, the Status of the Species, the Environmental Baseline, and Cumulative Effects, we have determined that the proposed action is not likely to jeopardize the continued existence of the Kemp's ridley sea turtle, loggerhead sea turtle (NWA DPS), green sea turtle (NA and SA DPSs), or leatherback sea turtle. These analyses focused on the impacts to, and population responses of, these species. Because the proposed action will not appreciably reduce the likelihood of survival and recovery of these species, it is our Opinion that the proposed action is also not likely to jeopardize the continued existence of any of these species.

In addition, after reviewing the current status of GSCH in Unit 8, the Environmental Baseline, the Effects of the Proposed Action, and the Cumulative Effects, it is our Opinion that the proposed action will not reduce the ability of designated critical habitat to support Gulf sturgeon



conservation, despite permanent adverse effects to an essential feature. Given the nature of the proposed action and the information provided above, we conclude that the action, as proposed, is likely to adversely affect, but is not likely to destroy or adversely modify, Unit 8 of GSCH.

## 9 INCIDENTAL TAKE STATEMENT

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Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. *Take* is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. *Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the reasonable and prudent measures (RPMs) and the terms and conditions (T&C) of the incidental take statement (ITS) of the Opinion. NMFS must estimate the extent of take expected to occur from implementation of the proposed action to frame the limits of the take exemption provided in the ITS. These limits set thresholds that, if exceeded, would be the basis for reinitiating consultation. The following section describes the extent of take that NMFS anticipates will occur as a result of the proposed action.

NMFS anticipates the total lethal take over the next 150 years as a result of the project will consist of up to 1,316 Kemp's ridley sea turtles, 43 loggerhead sea turtles (NWA DPS), 19 green sea turtles (NA DPS), 1 green sea turtles (South Atlantic DPS), and 1 leatherback sea turtle (Table 12). Based on the best available data, we do not anticipate any non-lethal take of the species listed above. The level of takes occurring annually is highly variable and influenced by sea temperatures, species abundances, monofilament accumulation, and other factors that cannot be predicted. Because one of the purpose of an ITS is to serve as a reinitiation trigger that provides clear signals that the level of anticipated take has been exceeded and, therefore, would require reexamination of the proposed action through a reinitiated consultation, we express the anticipated future take by species over the course of life of the project. The take numbers during the first 25 years, first 100 years, and 150 years are from Table 11. The take for the first 50 years and 75 years are calculated by dividing the take for the first 100 years by 75 (the years of reef maturity at year 100), and then multiplying the result by the number of years the reef has been mature (i.e., a 50 year reef has been mature for 25 years, and 75 year reef has been mature for 50 years). The resulting numbers were rounded down in order to be conservative for each species for the purpose of triggering reinitiation. The exceedance of any take estimate provided in Table 12 for any defined time period will require reinitiation.

**Table 12. Anticipated Future Take by Species and Distinct Population Segment (DPS) Over 150 Years**

<b>Sea Turtles</b>	<b>Estimated Lethal Take During First 25 Years</b>	<b>Estimated Lethal Take During First 50 Years</b>	<b>Estimated Lethal Take During First 75 Years</b>	<b>Estimated Lethal Take During First 100 Years</b>	<b>Estimated Lethal Take Over Entire 150 Years</b>
Kemp's ridley	0	358	716	1,074	1,316
Loggerhead N.W. Atlantic DPS	0	11	22	34	43
N. Atlantic Green DPS	0	5	10	15	19
S. Atlantic Green DPS	0	1	1	1	1
Leatherback	0	1	1	1	1

If any takes of species under NMFS's purview are observed taken during the proposed action authorized using this Opinion as the Section 7 consultation, it shall be immediately reported to [takereport.nmfsser@noaa.gov](mailto:takereport.nmfsser@noaa.gov) (include Opinion issue date, and the NMFS ECO identifier number [SERO-2019-03255]).

In order to monitor the impact of incidental take, the USACE must report the progress of the action and its impact on the species to NMFS as specified in the ITS (50 CFR § 402.14(i)(3)).

### **9.1 Effect of the Take**

NMFS has determined that the anticipated incidental take is not likely to jeopardize the continued existence of any species or DPS of ESA-listed sea turtle.

## **10 REASONABLE AND PRUDENT MEASURES**

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Section 7(b)(4) of the ESA requires NMFS to issue a statement specifying the impact of any incidental take on listed species, which results from an agency action otherwise found to comply with Section 7(a)(2) of the ESA. It also states the Reasonable and Prudent Measures (RPMs) necessary to minimize the impacts of take and the T&Cs to implement those measures, must be provided and must be followed to minimize those impacts. RPMs are measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02). Only incidental take by the federal agency that complies with the specified T&Cs is authorized.

The RPMs and T&Cs are specified as required, by 50 CFR 402.14(i)(1)(ii) and (iv), to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. The USACE has a continuing duty to regulate the activity covered by this ITS. If the USACE fails to adhere to the T&Cs through enforceable terms, and/or fails to retain oversight to ensure compliance with these T&Cs, the protective coverage of Section 7(o)(2) may lapse.

NMFS has determined that the following RPM must be implemented by the USACE (directly or through mandatory conditions of its authorization for the action):

- 1) The USACE will have measures in place to monitor and report all interactions with any protected species resulting from the proposed action.

## **11 TERMS AND CONDITIONS**

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In order to be exempt from the prohibitions of section 9 of the ESA, the Federal action agency must comply (or must ensure that any applicant complies) with the following T&Cs, which implement the above RPM:

- 1) If the permittee discovers or observes any live, damaged, injured or dead individual of an endangered or threatened species during construction or monitoring, the Permittee shall immediately notify the USACE, Mobile District Engineer so that any necessary stranding response coordination can be initiated with the U.S. Fish and Wildlife Service and/or National Marine Fisheries Service.
- 2) The federal action agency must ensure that the applicant reports all known takes of ESA-listed species to the NMFS SERO PRD. When the applicant becomes aware of any known reported capture, entanglement, stranding, or other take, the applicant must notify NMFS SERO PRD by email: [takereport.nmfs@noaa.gov](mailto:takereport.nmfs@noaa.gov).
  - a. Emails must reference this Opinion by the NMFS tracking number (SERO-2019-00452 Mississippi Gulf Fishing Banks Artificial Reefs) and date of issuance.
  - b. The email must state the species, date and time of the incident, general location and activity resulting in take, condition of the species (i.e., alive, dead, sent to rehabilitation), size of the individual, behavior, identifying features (i.e., presence of tags, scars, or distinguishing marks), and any photos that may have been taken.

## **12 CONSERVATION RECOMMENDATIONS**

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Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations identified in Biological 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 recommendation is a discretionary measure that NMFS believes is consistent with this obligation and therefore should be carried out by the federal action agency:

- 1) The permittee shall attempt annual monitoring and reef clean-ups as frequently as possible to prevent the accumulation of monofilament.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, NMFS requests notification of the implementation of any additional conservation recommendations.

## 13 REINITIATION OF CONSULTATION

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This concludes NMFS's formal consultation on the proposed action. As provided in 50 CFR 402.16, reinitiation of consultation is required where discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action on listed species or designated critical habitat in a manner or to an extent not considered in this Opinion, (3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this Opinion, or (4) a new species is listed or critical habitat designated that may be affected by the action. Pursuant to 50 CFR 402.14(i)(5), any taking which is subject to an ITS and which is compliance with the T&Cs is not a prohibited taking under the ESA.

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