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

F/SER31:SG
SERO-2023-00489

10/31/2023

Kelly Bunting
Project Manager, Panama City Permits Section
Jacksonville District Corps of Engineers
Department of the Army
415 Richard Jackson Boulevard, Suite 411
Panama City Beach, Florida 32407

Ref.: SAJ-1995-07320, City of Mexico Beach, Sandy Reef Artificial Reef, Mexico Beach, Bay County, Florida

Dear Kelly Bunting,

The enclosed Biological Opinion responds to your request for consultation with us, the National Marine Fisheries Service (NMFS), pursuant to Section 7 of the Endangered Species Act of 1973, as amended (16 U.S.C. § 1531 et seq.) for the above referenced action. The Opinion has been given the NMFS tracking number SERO-2023-00489. Please use the NMFS tracking number in all future correspondence related to this action.

The Opinion considers the effects of the U.S. Army Corp of Engineer's (USACE) proposal to authorize the expansion of an existing offshore artificial reef site and the deployment of artificial reef materials by the City of Mexico Beach in Bay County, Florida, on the following listed species: green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), leatherback sea turtle, hawksbill sea turtle, Gulf sturgeon, smalltooth sawfish (U.S. DPS) and giant manta ray. The Opinion is based on information provided by the USACE, the City of Mexico Beach, and the published literature cited within. NMFS concludes that the proposed action is not likely to adversely affect hawksbill sea turtle, Gulf sturgeon, the smalltooth sawfish (U.S. DPS), and giant manta ray. NMFS concludes that the proposed action is likely to adversely affect, but is not likely to jeopardize the continued existence of, green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and leatherback sea turtle.

We believe the proposed action will have No Effect on the South Atlantic DPS of green sea turtle. Limited information previously indicated that benthic juveniles from both the North Atlantic and South Atlantic DPSs may be found in waters off the mainland United States. However, additional research has determined that juveniles from the South Atlantic DPS are not likely to occur in these waters, including the action area for this project. Based on the foregoing, we believe green sea turtles from the South Atlantic DPS are not present within the action area.

NMFS is providing an Incidental Take Statement with this Opinion. The Incidental Take Statement describes Reasonable and Prudent Measures that NMFS considers necessary or appropriate to minimize the impact of incidental take associated with this action. The Incidental

Take Statement also specifies Terms and Conditions, including monitoring and reporting requirements with which the USACE and applicant must comply, to carry out the Reasonable and Prudent Measures.

We look forward to further cooperation with you on other projects to ensure the conservation of our threatened and endangered marine species and critical habitat. If you have any questions regarding this consultation, please contact Sarah Garvin, Consultation Biologist, by phone at (727) 342-0249, or by email at Sarah.Garvin@noaa.gov.

Sincerely,

STRELCHECK.ANDREW.JAMES.1365863152 Digitally signed by STRELCHECK.ANDREW.JAMES.1365863152 Date: 2023.10.31 14:32:26 -04'00'

Andrew J. Strelcheck
Regional Administrator

Enclosure (s):
NMFS Biological Opinion SERO-2023-00489
cc: Kelly.A.Bunting@usace.army.mil
nmfs.ser.esa.consultations@noaa.gov
File: 1514-22.f.4

Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	U.S. Army Corps of Engineers Permit number: SAJ-1995-07320
Applicant:	The City of Mexico Beach
Activity:	Authorizing the Expansion of an Artificial Reef Site and the Deployment of Artificial Reef Materials
Location:	Mexico Beach, Bay County, Florida
Consulting Agency:	National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida NMFS Tracking Number: SERO-2023-00489

Approved by:

STRELCHECK.ANDREW.JAMES.1365863152 Digitally signed by STRELCHECK.ANDREW.JAMES.1365863152 Date: 2023.10.31 14:32:57 -04'00'

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

Date Issued: 10/31/2023

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ACRONYMS, ABBREVIATIONS, AND UNITS OF MEASURE

ac	acre(s)
BMP	Best Management Practice
BOEM	Bureau of Ocean Energy Management
°C	degrees Celsius
CCL	Curved Carapace Length
CFR	Code of Federal Regulations
cm	centimeter(s)
DDT	Dichlorodiphenyltrichloroethane
DEIS	Draft Environmental Impact Statement
DPS	Distinct Population Segment
DTRU	Dry Tortugas Recovery Unit
DWH	Deepwater Horizon
ECO	Environmental Consultation Organizer
EIS	Environmental Impact Statement
EPA	Environmental Protection Agency
ESA	Endangered Species Act of 1973, as amended (16 U.S.C. § 1531 et seq.)
°F	degrees Fahrenheit
FAD	Fish Aggregating Device
FERC	Federal Energy Regulatory Commission
FMP	Fishery Management Plan
ft	foot/feet
FR	Federal Register
ft^2	square foot/feet
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
GADNR	Georgia Department of Natural Resources
GCRU	Greater Caribbean Recovery Unit
GPS	Global Positioning System
in	inch(es)
km	kilometer(s)
m	meter(s)
mi	mile(s)
mi^2	square mile(s)
MLLW	Mean Lower Low Water
MMPA	Marine Mammal Protection Act
NAD 83	North American Datum of 1983
NEFSC	NMFS Northeast Fisheries Science Center
NGMRU	Northern Gulf of Mexico Recovery Unit
nm	nautical mile(s)
nm^2	square nautical mile(s)
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
Opinion	Biological Opinion
PCB	Polychlorinated Biphenyls
PDC	Project Design Criteria

PFC	Perfluorinated Chemicals
PFRU	Peninsular Florida Recovery Unit
PRD	NMFS Protected Resources Division
SAV	Submerged Aquatic Vegetation
SCDNR	South Carolina Department of Natural Resources
SCL	Straight Carapace length
SERO	NMFS Southeast Regional Office
SEFSC	NMFS Southeast Fisheries Science Center
STSSN	Sea Turtle Stranding and Salvage Network
TED	Turtle Exclusion Device
TEWG	Turtle Expert Working Group
U.S.	United States
USACE	United States Army Corps of Engineers
USCG	United States Coast Guard
USFWS	United States Fish and Wildlife Service

1 INTRODUCTION

1.1 Overview

Section 7(a)(2) of the ESA, requires that each federal agency ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. Section 7(a)(2) requires federal agencies to consult with the appropriate Secretary in carrying out these responsibilities. The NMFS and the USFWS share responsibilities for administering the ESA. Consultations on most ESA-listed marine species and their critical habitat are conducted between the federal action agency and NMFS (hereafter, may also be referred to as we, us, or our).

Consultation is required when a federal action agency determines that a proposed action "may affect" ESA-listed species or critical habitat and can be conducted informally or formally. Informal consultation is concluded after NMFS issues a Letter of Concurrence that concludes that the action is "not likely to adversely affect" ESA-listed species or critical habitat. Formal consultation is concluded after we issue a Biological Opinion (hereafter, referred to as an/the Opinion) that identifies whether a proposed action is "likely to jeopardize the continued existence of an ESA-listed species" or "destroy or adversely modify critical habitat," in which case Reasonable and Prudent Alternatives to the action as proposed must be identified to avoid these outcomes. An Opinion often states the amount or extent of anticipated incidental take of ESA-listed species that may occur, develops Reasonable and Prudent Measures necessary to minimize the impacts, i.e., amount or extent, of the anticipated incidental take, and lists the Terms and Conditions to implement those measures. An Opinion may also develop Conservation Recommendations that help benefit ESA-listed species

This document represents NMFS's Opinion based on our review of potential effects of the USACE's proposal to authorize the expansion of an existing offshore artificial reef site and the deployment of artificial reef materials by the City of Mexico Beach (the applicant) in Bay County, Florida on the following listed species: green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), leatherback sea turtle, hawksbill sea turtle, Gulf sturgeon, smalltooth sawfish (U.S. DPS), and giant manta ray. Our Opinion is based on information provided by the USACE, the applicant, and the published literature cited within.

On July 5, 2022, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR part 402 in 2019 ("2019 Regulations," see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order. On November 14, 2022, the Northern District of California issued an order granting the government's request for voluntary remand without vacating the 2019 regulations. The District Court issued a slightly amended order two days later on November 16, 2022. As a result, the 2019 regulations remain in effect, and we are applying the 2019 regulations here. For purposes of this consultation and in an abundance of caution, we considered whether the substantive analysis and conclusions articulated in the Opinion and

Incidental Take Statement would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

1.2 Consultation History

The following is the consultation history for the NMFS ECO tracking number SERO-2023-00489 Mexico Beach Sandy Reef Artificial Reef.

On December 3, 2021, we issued a letter of concurrence to NOAA Restoration Center on the Artificial Reef Creation and Restoration Project proposed for funding under the DWH Oil Spill Natural Resource Damage Assessment in the Florida Trustee Implementation Group (SERO-2021-02759, DWH NRDA FL TIG RP#2-Artificial Reefs). The Mexico Beach Sandy Reef Artificial Reef (SAJ-1995-07320) was included in this consultation. The use of high-relief artificial reef materials was not proposed for any of the sites funded by NOAA Restoration Center. In our Letter of Concurrence, we determined that the proposed project would not adversely affect any ESA-listed species or designated critical habitat.

On April 18, 2023, we received a written request for expedited informal consultation under Section 7 of the ESA from the USACE to authorize the expansion of an existing offshore artificial reef site and the deployment of high-relief artificial reef materials (i.e., materials greater than 7 ft in height from the seafloor) by the City of Mexico Beach (the applicant) in Bay County, Florida.

On June 15, 2023, we informed the USACE of the need for formal consultation because of the applicant's proposed use of high-relief artificial reef materials for artificial reef creation. The previous consultation with NOAA Restoration Center (SERO-2021-02759, DWH NRDA FL TIG RP#2-Artificial Reefs, issued December 3, 2021) did not consider the deployment of high-relief artificial reef materials. We also requested additional information related to the project description, BMPs, and mitigation measures.

We received a final response from USACE on July 12, 2023, and initiated formal consultation that day.

2 PROPOSED ACTION

2.1 Project Details

2.1.1 Project Description

The USACE proposes to authorize the expansion of the existing Sandy Reef artificial reef deployment area located offshore of the City of Mexico Beach, Bay County, Florida. The USACE also proposes to authorize the deployment of artificial reef materials into the Gulf of Mexico by the City of Mexico Beach. The purpose of the proposed project is to enhance marine habitat and opportunities for recreational fishing and diving. The expanded reef area will measure approximately 0.98 nm^2 (1.34 nm x 0.73 nm) and cover 821 ac. The existing reef area currently measures 0.5 nm x 0.5 nm and covers 220 ac.

Materials will be deployed opportunistically within the reef area as funding and materials become available. The applicant estimates a maximum of approximately 5 total deployments per year in the expanded reef area, for a total of 50 deployments during the ten year duration of the permit.

Materials proposed for deployment include:

1) Designed and engineered concrete reef modules manufactured specifically for use as artificial reef habitat. These prefabricated reef units are constructed of concrete, rock and steel;
2) Heavy gauge metal materials with a thickness of ¼ in or greater and weighing at least 500 lbs. Such materials may be metal prefabricated reef units, solid steel beams, or low profile steel pipes;
3) Secondary use concrete materials weighing at least 500 lbs such as concrete culverts, concrete bridge materials, or other similar concrete materials;
4) Concrete and steel bridge materials with a thickness of ¼" or greater and weighing at least 500 lbs;
5) Natural rock or limestone boulders weighing at least 500 lbs each; and
6) Heavy gauge ferrous and aluminum alloy metal-hulled vessels that equal or exceed 60 ft hull length.

The proposed project also includes the deployment of metal-hulled vessels that equal or exceed 60 ft hull length. 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 ft or more from the seafloor and that has a footprint greater than 200 ft^2 (individually or collectively), excluding prefabricated artificial reef modules.

The applicant proposes a minimum 40 ft clearance. The maximum profile of any reef material, including vessels, would range from 3 ft to 60 ft. Water depths within the reef area range between a minimum of 94 ft and maximum of 100 ft at MLLW. Further, the top elevation of the reef will not be less than 6 ft below the water surface.

Artificial reef construction utilizes a number of vessels for material deployment. Reef deployments typically use a combination of barges, ships with a mounted crane, and small tugboats. Vessel speeds will vary and are dependent on contractor selection and type of vessel being used. Vessel speeds will be reduced while maintaining sufficient maneuverability and navigation. The exact travel routes to and from the proposed reef area will be restricted to the existing navigation channels originating in Port St. Joe, Mexico Beach Canal, and St. Andrews Pass, and the Gulf of Mexico. An estimated maximum of 40 vessel loads/trips associated with reef deployments are anticipated. The time window of operations will be during daylight hours anytime throughout the year, but will depend on favorable weather, sea conditions, and material funding and availability. Time underway for each vessel will depend on the port of call. If coming from Orange Beach, Alabama, it will be a full day of operation. If leaving from Mexico Beach, it will likely be a half day of operation.

Materials will be transported to the reef deployment area by barge or ships with mounted cranes.

Cranes or backhoes or similar heavy machinery will be used to drop reef materials directly into the water. No materials will be deployed within a 300 ft buffer zone inside the boundaries of the reef area. Pre- and post-deployment dive surveys will be conducted to ensure no seagrass or live or hardbottom are present and to ensure proper deployment of reef materials. Pre-deployment surveys will confirm the presence of sandy bottom. Any areas of exposed live bottom habitat will be marked with buoys and avoided during deployment. If any live bottom is encountered, a minimum buffer of 200 ft from deployment activities will be maintained and no new reef materials will be placed within this buffer zone from live bottom habitat. Temporary buoys or markers will be deployed immediately prior to the deployment of any reef materials. These buoys or markers will be removed following completion of deployment.

The proposed action includes the potential deployment of vessels of opportunity that may become available to the applicant during the life of the permit. Vessel profiles would be restricted to ensure the proposed 40-ft minimum clearance. Vessels would not be deployed until all necessary inspections and clearances have been obtained or waived and a stability analysis has been completed demonstrating that the vessel would be stable during a 50-year storm event based on vessel and deployment site characteristics. The applicant would follow the national guidance regarding preparation of vessels for deployment as artificial reefs which are available at: https://www.epa.gov/sites/default/files/2015-09/documents/artificialreefguidance.pdf.

Vessels will be modified to prevent entrapment of sea turtles, mammals, and divers. Ingress and egress points would be of sufficient size to prevent entrapments. Vessels will be inspected by FWC to identify and correct entrapment hazards prior to deployment of any vessels. The applicant has agreed to abide by the following special conditions if a vessel becomes available to be deployed within the reef site, as recommended by FWC. Deployment of any vessel as artificial reef material is prohibited unless written authorization has first been obtained from the USACE. The applicant must first submit a project-specific deployment plan to the USACE to request such authorization to deploy vessels, and the plan must include the following information:

a. Detailed description of the proposed deployment including vessel material type, deployment depth, intended orientation of the vessel (e.g., upright, on its side, upside down), navigational clearance with the material in all orientations, and the weight and dimensions of the selected vessel;
b. Stability analysis of the proposed vessel at the depth and location proposed for deployment;
c. A pre-deployment preparation plan describing how compliance with the EPA and MARAD "National Guidance: Best Management Practices for Preparing Vessels Intended to Create Artificial Reefs" has been followed. This document is available at: https://www.epa.gov/sites/production/files/2015-09/documents/artificialreefguidance.pdf;
d. Tow and anchoring plan describing how the vessel will be towed to and anchored at the deployment site;
e. Sink Plan describing the methods used to deploy the vessel. If the use of explosives is requested, a detailed explosive and detonation plan including justification documenting the need for the use of explosives as the only deployment alternative, and a marine mammal monitoring and vessel security plan must be provided;
f. A monitoring plan describing on-water and pre-deployment monitoring, immediate post-

deployment monitoring, and annual monitoring activities to document that the vessel is deployed/located within the permitted area including coordinates, orientation of vessel (e.g., upright, side, upside down), structural integrity status (i.e., is the vessel in one piece, are pieces being disassociated from the structure), and documentation that the vessel meets USCG navigational requirements;

g. A copy of the proposed project budget, and documentation that the permittee has approved funding adequate to procure, clean, deploy, and conduct monitoring of the selected vessel once it has been deployed.

2.1.2 Mitigation Measures

The following construction conditions and project design criteria (PDC) will be implemented during deployments to avoid and minimize potential effects to ESA-listed species and their habitats.

- **Planning and Deployment Guidelines.** The applicant will incorporate the following guidelines when planning for and deploying artificial reefs:
    - *ASMFC/GSMFC Guidelines for Marine Artificial Reef Materials,*
    - *EPA's National Guidance: Best Management Practices (BMPs) for Preparing Vessels Intended to Create Artificial Reefs,* and
    - *NOAA/NMFS National Artificial Reef Plan*.
- **Initial Agency Notification.** The Permittee shall provide to the USACE, NOAA, and USCG written notification of the planned deployment start date at least 2 weeks prior to the initial deployment on the authorized artificial reef site.
- **Pre-Deployment Notification:** No less than 14 days prior to deployment of material on an artificial reef, the applicant shall transmit by email a complete and signed *"Florida Artificial Reef Materials Cargo Manifest and Pre-Deployment Notification"* form to the USACE and FWC to allow inspection of the proposed reef materials as deemed necessary by the agencies.
    - Inspection is allowable at the staging area.
    - By signing the Pre-Deployment Notification the applicant certifies all materials are free from asphalt, petroleum, other hydrocarbons and toxic residues. The applicant shall not deploy material if notified by the USACE or FWC that the material is questionable. The material needs to be evaluated before it is released for deployment.
    - Any material deemed unacceptable for reef material will be disposed in an approved upland disposal site.
    - Deployment of the material shall not occur until after the end of the 14-day inspection period.
    - The applicant shall ensure both a copy of the permit and the signed *"Florida Artificial Reef Materials Cargo Manifest and Pre-Deployment Notification"* form are maintained aboard the deployment vessel at all times during loading, transit, and deployment.
    - The applicant shall provide a record of all inspections, clearances or waivers to the USACE along with the pre-deployment notification.

- **Protected Species Construction Conditions.** The applicant will comply with NMFS SERO's "*Protected Species Construction Conditions,*" dated May 2021.
- **Vessel Strike Avoidance Measures.** The applicant will comply with NMFS SERO's "*Vessel Strike Avoidance Measures,*" dated May 2021, for species protected under the ESA and the MMPA. In particular, the applicant will ensure the following measures will be implemented:
    - All vessels associated with the project shall operate at "idle/no wake" 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, and in all depths after a protected species has been observed in and has recently departed the area.
    - All vessels will follow deep-water routes (e.g., marked channels) whenever possible.
    - The applicant shall instruct all personnel associated with the project of the potential presence of protected species and any critical habitat in a vessel transit area, and the need to avoid collisions with them. All vessels should have personnel onboard responsible for observing water-related activities for the presence of these species.
    - If a protected species is sighted, attempt to maintain a distance of 150 ft or greater between the animal and the vessel, and reduce speed and avoid abrupt changes in direction until the animal(s) have left the area.
    - If a protected species is sighted within 300 ft of the vessel, all appropriate precautions shall be implemented to avoid a collision. These precautions shall include cessation of any vessel movement when a protected species is observed within 150 ft of operations (excluding at times when movement is required for safe navigation [e.g., transiting inlets]). Operation may not resume until the protected species has departed the immediate area of its own volition.
- **Daylight Hours.** All artificial reef work will only take place during reasonably calm, clear weather and during daylight hours.
- **Benthic Survey.** No artificial reef materials shall be deployed until a benthic assessment of the bottom conditions has been accomplished by diver or submersible video camera. The inspection of the deployment area may occur at the time of deployment, but no more than 1 year prior to deployment.
- **Buffers.** Siting of artificial reef materials may not occur within the following buffers.
    - Siting of any vessel, aircraft, or large and high-relief material (e.g., bridge spans) may not occur within 1,500 ft of any documented coral colonies. Any vessel used in the deployment of an artificial reef may not anchor or moor within 1,500 ft of any documented coral colonies.
    - The applicant shall maintain a deployment buffer of at least 200 ft from any other submerged aquatic resources, including seagrasses, macroalgae, sponges, and oysters, when placed in areas of sand. If materials are off-loaded from a barge or placed in areas that may generate turbidity (e.g., areas with fines or muck), a 500 ft buffer is required.
    - No artificial reef material will be deployed in any area within 1,100 ft off any identified sea turtle nesting beach that predominantly consists of sandy benthic habitat.

- **Removal of Non-essential Structures.** All railings and other non-essential structures that could otherwise easily accumulate monofilament line should be removed from all high-relief materials.
- **Preparation of Vessels for Deployment.** Pursuant to the EPA BMPs, thorough preparation and cleaning is required before vessels may be used for reefs. Military surplus and vessel structures such as ladders, rails, booms, antennas, etc. will be removed to reduce the potential accumulation of abandoned fishing tackle and lines.
- **Decontamination**. 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.
- **Weight requirements.** No individual artificial reef component (i.e., prefabricated module, concrete piece, etc.) will weigh less than 500 lbs, with the exception of materials deployed directly by authorized county or state programs in low-energy environments (e.g., Reef Ball "Bay Ball" or "Mini-Bay Ball" in shallow estuaries or bays). All materials shall be of sufficient weight in-water to not move from the site post-deployment.
- **Entanglement Prevention.** Reef structures, materials, and installation methods shall be designed and deployed to prevent entanglement and entrapment of listed species. Open-bottom prefabricated artificial reef modules may not be deployed unless the module also has an opening at the top that is sufficient to allow the escapement of an adult loggerhead sea turtle. For an open-bottom artificial reef module that is triangular (e.g., pyramid) or square, the top must be open and each of the side's exposed opening edges (i.e., top edge) must be at least 4 ft long. Optionally, a triangular (e.g., pyramid) open-bottom artificial reef module may reduce the length of two of the side's exposed opening edges (i.e., top edge) to a minimum of 3 ft long if the third side is lowered to allow a 4 ft length opening edge on that third side. For instance, this would require a pyramid module with a 10 ft base that is 8 ft high to be cut down and remove 2.4 ft of material on two sides and 3.2 ft of material on the third side to produce the required opening. Open-bottom prefabricated modules with a round or oval opening at the top must have a diameter of at least 4 ft as measured from any two points along the exposed opening edge.
- **Egress.** Open-bottom fabricated artificial reef modules may not include any additional sub-components or other material within the interior or obstructing the top opening that could impair the egress of a sea turtle.
- **Protrusions.** For all secondary-use, recycled concrete and similar materials, all steel reinforcement rods, rebar, and other protrusions must be cut at the base of the concrete and level with the surface concrete so that no metal protrudes from the concrete's surface.
- **FADs.** Mid-water fish aggregating devices (FADs) will not be used.
- **Explosives.** Explosives will not be used to deploy artificial reefs.
- **Protected Species Sightings.** Deployment activities will not commence until the project supervisor reports that no sea turtles, marine mammals, or other ESA-listed species have been sighted within 150 ft (50 yds) of the active deployment site (i.e., barge carrying material or moored vessel to be scuttled [i.e., deliberately sunk]) for at least 20 minutes. Deployment activities will cease immediately if sea turtles, marine mammals, or other ESA-listed species are sighted within 150 ft (50 yds) of the active deployment site. Deployment activities will not recommence until the project supervisor reports that no

sea turtles, marine mammals, or other ESA-listed species have been sighted for at least 20 minutes.

- **Reporting.** Any collision with or injury to an ESA-listed species shall be reported immediately to the NMFS SERO's Endangered Species Take Report Form. For additional reporting resources, please go to: https://www.fisheries.noaa.gov/report.

2.1.3 Best Practices

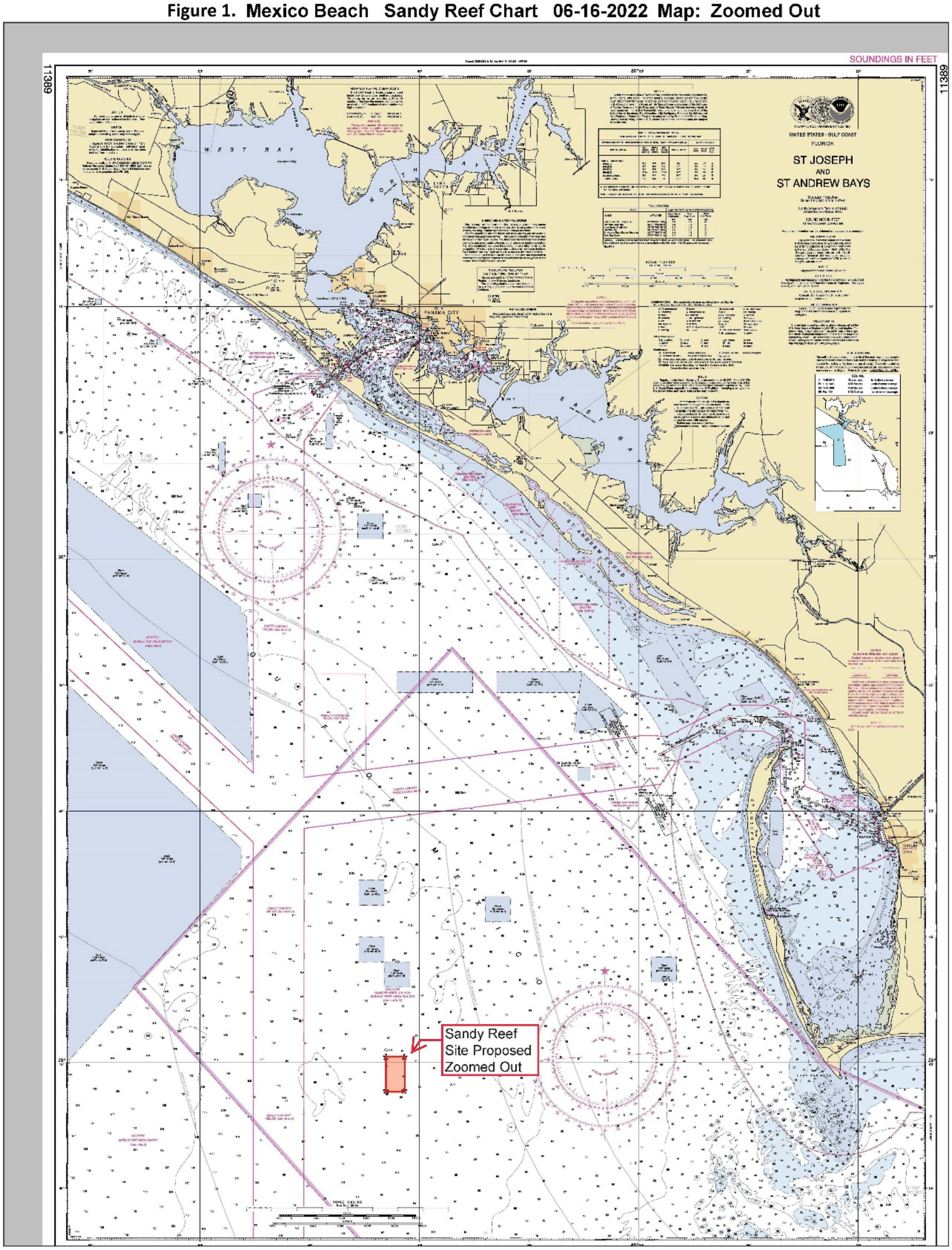
The following best practices will be implemented following completion of the project to avoid and minimize potential effects to ESA-listed species and their habitats.

- **Post-Deployment Notification**: No less than 30 days following each deployment, the applicant will complete and submit the "*Florida Artificial Reef Materials Placement Report and Post- Deployment Notification*" form to the USACE, FWC, and NOAA.
- **Violation of Reef Parameters Notification:** In the event reef material is deployed in a location or manner contrary to the Reef Parameters Special Condition, which requires that all reef materials be deployed within the reef area boundaries and with a minimum clearance of 40 ft from the top of the deployed material relative to MLLW, the applicant shall immediately notify the USCG Station and provide information as requested by the station. The applicant shall notify NOAA, USCG, and USACE in writing within 24 hours of the occurrence. At a minimum, the written notification shall explain how the deployed material exceeds the authorized reef parameters, a description of the material, a description of the vessel traffic in the area, the deployment location in nautical miles at compass bearing from obvious landmarks, the location of the unauthorized material in latitude and longitude coordinates (degree, minute, decimal minute format to the third decimal place), and the water depth above the material from MLW. The document will list the information provided by telephone to the USCG as noted above and include the time of the call and the name of the USCG personnel receiving the information.
- **Annual Monitoring**: The applicant will conduct yearly monitoring within the Sandy Reef deployment area. Specifically, within 12 months of the effective date of the permit, and every 12 months thereafter for the duration of the permit, the applicant will submit a report summarizing deployments and issues associated with the reef in the preceding 12 months to both the USACE and FWC. The report will document any known changes in material condition (stability, durability, and location) as compared to those same characteristics at the time of deployment. The report may include, but is not limited to, use trends, site management constraints and resolutions, management techniques, modifications of operations, plans, and lessons learned. The report must also include results of any performance monitoring (description of fish and other biota observed).

2.2 Action Area

The existing Sandy Reef artificial reef area is located in the Gulf of Mexico, approximately 20.9 nm offshore Mexico Beach, Bay County, Florida. The centroid coordinates of the project site are 29.665278°, -85.687556° (NAD 83; see Figure 1).

Figure 1. Mexico Beach Sandy Reef Chart 06-16-2022 Map: Zoomed Out



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(Not For Navigational Use!)

Figure 1. Location of the proposed expanded Sandy Reef artificial reef area in the Gulf of Mexico relative to the shoreline of Mexico Beach, Bay County, Florida (image provided by USACE).

The USACE originally authorized the creation of the existing Sandy Reef artificial reef area on January 22, 1996. That authorization expired on January 19, 2000. The existing site was authorized as a ½ nm by ½ nm square area, or approximately 220 acres (Figure 3 depicts the existing site). Side-scan surveys of the proposed expanded reef area were conducted by FWC in December 2016, and underwater surveys were conducted by the Mexico Beach Artificial Reef Association (MBARA) in November 2016 and July 2017. These surveys showed that the proposed expanded area is free of submerged aquatic vegetation and hardbottom or live bottom resources, and materials previously deployed are within the existing, permitted boundaries. The expanded reef area encompasses the remains of a sunken shrimp boat in the northwest corner. This sunken boat is currently charted on NOAA chart 11389. The boundary coordinates for the expanded reef area listed below in Table 1.

Table 1. Boundary Coordinates for Expanded Sandy Reef Area.

Boundary Waypoint	Latitude	Longitude	Water Depth Range
Northeast Corner	29.676389°	-85.680556°	94 to 100 ft
Northwest Corner	29.676389°	-85.694444°	
Southwest Corner	29.654167°	-85.694444°	
Southeast Corner	29.654167°	-85.680556°	

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 federal action, the action area includes the 0.98 nm^2 area contained within the boundary of the expanded Sandy Reef artificial reef deployment area (also referred to as the project site) and the transit routes for deployment, support, and monitoring vessels.

3 EFFECTS DETERMINATIONS

Please note the following abbreviations are only used in **Table 1** and **Table 2** and are not, therefore, included in the list of acronyms: E = endangered; T = threatened; LAA = likely to adversely affect; NLAA = may affect, not likely to adversely affect.

3.1 Effects Determinations for ESA-Listed Species

3.1.1 Agency Effects Determinations

We have assessed the ESA-listed species that may be present in the action area and our determination of the project's potential effects is shown in **Table 2** below.

Table 2. ESA-listed Species in the Action Area and Effect Determinations

Species (DPS)	ESA Listing Status	Listing Rule/Date	Most Recent Recovery Plan (or Outline) Date	USACE Effect Determination	NMFS Effect Determination
Sea Turtles					

Species (DPS)	ESA Listing Status	Listing Rule/Date	Most Recent Recovery Plan (or Outline) Date	USACE Effect Determination	NMFS Effect Determination
Green sea turtle (North Atlantic DPS)	T	81 FR 20057/ April 6, 2016	October 1991	NLAA	LAA
Hawksbill sea turtle	E	35 FR 8491/ June 2, 1970	December 1993	NLAA	NLAA
Kemp's ridley sea turtle	E	35 FR 18319/ December 2, 1970	September 2011	NLAA	LAA
Leatherback sea turtle	E	35 FR 8491/ June 2, 1970	April 1992	NLAA	LAA
Loggerhead sea turtle (Northwest Atlantic DPS)	T	76 FR 58868/ September 22, 2011	December 2008	NLAA	LAA
Fishes					
Giant manta ray	T	83 FR 2916/ January 22, 2018	2019 (Outline)	NLAA	NLAA
Gulf sturgeon (Atlantic sturgeon, Gulf subspecies)	T	56 FR 49653/ September 30, 1991	September 1995	NLAA	NLAA
Smalltooth sawfish (U.S. DPS)	E	68 FR 15674/ April 1, 2003	January 2009	NLAA	NLAA

Unlike the other ESA-listed sea turtles, hawksbill sea turtles are not likely to be adversely affected by the proposed action. The most recent STSSN data available for Zone 8 (2007-2016), which includes the action area, shows 1 reported stranding of hawksbill sea turtle in offshore Bay County. The cause of death is unknown, and the animal exhibited oil/tar contamination. Subsequently, we believe the presence of hawksbill sea turtles within the action area will be rare, and it is extremely unlikely they would be found interacting with artificial reef material. Therefore, NMFS believes that the proposed action may affect, but is not likely to adversely affect the hawksbill sea turtle as opposed to the other ESA-listed sea turtles that are present within the action area. Hawksbill sea turtles will not be discussed further in this Opinion and further references to ESA-listed sea turtles will be limited to green sea turtles (North Atlantic DPS), Kemp's ridley sea turtles, loggerhead sea turtles (Northwest Atlantic DPS), and leatherback sea turtles.

3.1.2 Effects Analysis for ESA-Listed Species Not Likely to be Adversely Affected by the Proposed Action

ESA-listed sea turtles, smalltooth sawfish, Gulf sturgeon, and giant manta ray may be adversely affected by their inability to access the project sites for foraging, refuge, and nursery habitat due to their avoidance of construction activities and related noise. We determined these effects are 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 5 days per deployment opportunity), and construction will be limited to daylight hours only. Species will be able to move around the project sites once deployment is complete and at night.

ESA-listed sea turtles, smalltooth sawfish, Gulf sturgeon, and giant manta ray could be physically injured if struck by transport vessels or materials 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. Further, the PDCs require that deployment activities will cease immediately if any protected species is sighted within 150 ft (50 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 150 ft (50 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 150 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, or at least 20 minutes have passed since the animal was last seen.

ESA-listed species, namely sea turtles, smalltooth sawfish, Gulf sturgeon, and giant manta ray, may also 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 6, we believe entanglement in high-relief artificial reefs materials may adversely affect sea turtles.

Low-relief and 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 listed above in Section 2.1 would further reduce the likelihood of entanglement and entrapment. The PDCs for entanglement prevention require that materials of design, such as “reef balls” used for offshore deployments, are to have an opening at the top that is sufficient to allow the escapement of an adult loggerhead sea turtle. The PDCs for protrusions require 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, the PDCs for Protected Species Sightings requires that deployment activities will not commence until the project supervisor reports that no protected species have been sighted within 150 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 150 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 ESA-listed sea turtles are likely to experience entanglement events. We believe entanglement of the ESA-listed fish species that may be in the action area is extremely unlikely to occur because we have no information documenting any artificial reef entanglement event involving these fish species and because it is extremely unlikely that these species will utilize artificial reefs as habitat. Gulf sturgeon, smalltooth sawfish, and giant manta ray do not typically feed or rest on or 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 known to have adverse effects on sea turtles due to potential entanglement. Adverse effects from the proposed action on green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and leatherback sea turtle are discussed further in Section 6.

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. We believe the proposed action is extremely unlikely to increase the risk of incidental capture because there is no evidence that the establishment of artificial reefs increases the numbers of fishers or boats participating in a given fishery.

3.1.3 ESA-Listed Species Likely to be Adversely Affected by the Proposed Action

We have determined the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and leatherback sea turtle are likely to be adversely affected by the proposed action and thus require further analysis. We provide greater detail on the potential effects to these species from the proposed action in the Effects of the Action (Section 6.1) and whether those effects, when considered in the context of the Status of the Species (Section 4.1), the Environmental Baseline (Section 5), and the Cumulative Effects (Section 7), are likely to likely to jeopardize the continued existence of these ESA-listed species in the wild.

3.2 Effects Determinations for Critical Habitat

3.2.1 Agency Effects Determinations

The project is not located in critical habitat, and there are no potential routes of effect to any critical habitat.

4 RANGE WIDE STATUS OF ESA-LISTED SPECIES CONSIDERED FOR FURTHER ANALYSIS

4.1 Sea Turtles

There are 4 species of sea turtles considered further for analysis in this Opinion: green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS). All 4 species travel widely throughout the South Atlantic, Gulf of Mexico, and the Caribbean, and may be adversely affected by the proposed action. Section 4.1.1.1 of this Opinion will address the general threats that confront all sea turtle species. The remainder of Section 4.1.1 (Sections 4.1.2 – 4.1.5) will address information on the distribution, life history, population structure, abundance, population trends, and unique threats to each species of sea turtle further discussed in this Opinion.

4.1.1 General Threats Faced by All Sea Turtle Species

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

Fisheries
Incidental bycatch in commercial fisheries is identified as a major contributor to past declines, and threat to future recovery, for all of the sea turtle species (NMFS and USFWS 1991; NMFS and USFWS 1992; NMFS and USFWS 1993; NMFS and USFWS 2008; NMFS et al. 2011). Domestic fisheries often capture, injure, and kill sea turtles at various life stages. Sea turtles in the pelagic environment are exposed to U.S. Atlantic pelagic longline 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 (Section 4.1.1) for more specific information regarding federal and state managed fisheries affecting sea turtles within the action area). The Southeast U.S. shrimp fisheries have historically been the largest fishery threat to benthic sea turtles in the southeastern United States, and continue to interact with and kill large numbers of sea turtles each year.

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

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

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

Environmental Contamination
Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., DDT, PCB, and PFC), and others that may cause adverse health effects to sea turtles (Garrett 2004; Grant and Ross 2002; Hartwell 2004; Iwata et al. 1993). Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges may directly injure individuals through skin contact with oils (Geraci 1990), inhalation at the water's surface and ingesting compounds while feeding (Matkin and Saulitis 1997). Hydrocarbons also have the

potential to impact prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area.

The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (DWH Trustees 2015). Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in Sargassum algae mats in the convergence zones, where currents meet and oil is collected. Sea turtles found in these areas were often coated in oil and had ingested oil or both. 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 lost, abandoned or discarded fishing gear) as they feed along oceanographic fronts where debris and their natural food items converge. Marine debris can cause significant habitat destruction from derelict vessels, further exacerbated by tropical storms moving debris and scouring and destroying corals and seagrass beds, for instance. Sea turtles that spend significant portions of their lives in the pelagic environment (i.e., juvenile loggerheads, and juvenile green turtles) are especially susceptible to threats from entanglement in marine debris when they return to coastal waters to breed and nest.

Climate Change

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

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

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

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

Other Threats

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

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

4.1.2 Status of Green Sea Turtle – North Atlantic DPS

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 DPSs (81 FR 20057 2016) (Figure 2). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific DPSs were listed as threatened. Only individuals from the South Atlantic DPS and North Atlantic DPS may occur in waters under the purview of the NMFS SE Region, with South Atlantic DPS individuals only expected to occur in the U.S. Caribbean.

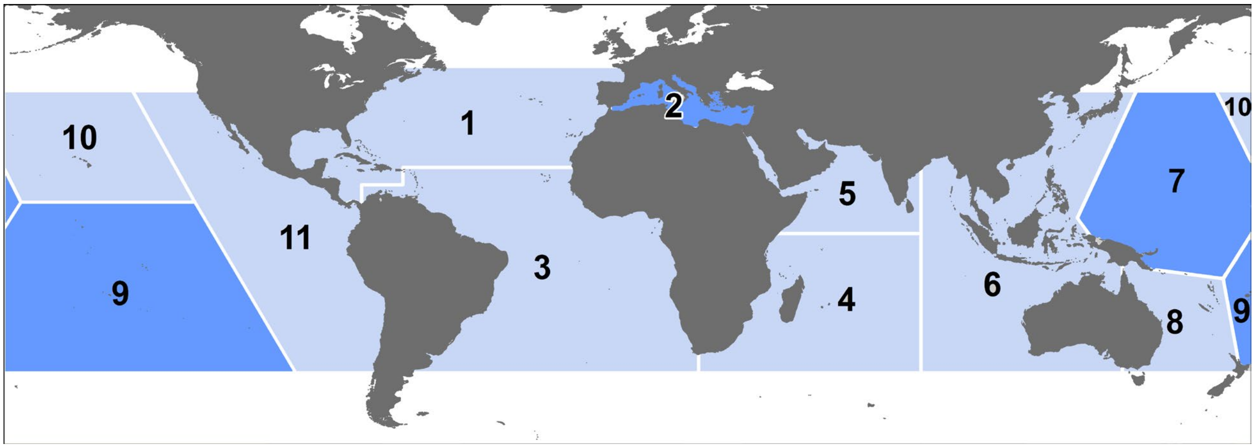

Figure 2. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11. 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 two largest nesting populations are found at Tortuguero, on the Caribbean coast of Costa Rica (part of the North Atlantic DPS), and Raine Island, on the Pacific coast of Australia along the Great Barrier Reef.

Differences in mitochondrial DNA properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; FitzSimmons et al. 2006). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Limited early information indicated that within U.S. waters benthic juveniles from both the North Atlantic and South Atlantic DPSs may be found on foraging grounds. Two small-scale studies provided an insight into the possible 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 South Atlantic 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 South Atlantic

DPS (Bass and Witzell 2000). 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). However, with additional research it has been determined that South Atlantic juveniles are not likely to be occurring in U.S. mainland coastal waters in anything more than negligible numbers. Jensen et al. (2013) indicated that the earlier studies might represent a statistical artifact as they lack sufficient precision, with error intervals that span zero. More recent studies with better rookery baseline representation found negligible (<1%) contributions from the South Atlantic DPS among Texas and Florida GoM juvenile green turtle assemblages (Shamblin et al. 2016, 2018). Finally, an as-yet published genetic analysis of samples from various coastal areas in the Gulf of Mexico and Atlantic has now solidified the conclusion that South Atlantic juveniles represent at best a negligible number of individuals in mainland United States waters (Peter Dutton, SWFSC, pers. comm. April 2022). Therefore, we will not consider South Atlantic DPS individuals when conducting consultations for projects in the waters off the mainland United States.

The North Atlantic DPS boundary is illustrated in Figure 2. Four regions support nesting concentrations of particular interest in the North Atlantic 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 North Atlantic DPS green sea turtles within the southeastern United States includes sandy beaches between Texas and North Carolina, as well as Puerto Rico (Dow et al. 2007; NMFS and USFWS 1991). The vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Principal U.S. nesting areas for green sea turtles are in eastern Florida, predominantly Brevard south through Broward counties.

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

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

After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. This early oceanic phase remains one of the most poorly understood aspects of green sea turtle life history (NMFS and USFWS 2007). Green sea turtles exhibit particularly slow growth rates of about 0.4-2 inches (1-5 cm) per year (Green 1993), which may be attributed to their largely herbivorous, low-net energy diet (Bjorndal 1982). At approximately 8-10 inches (20-25 cm) carapace length, juveniles leave the pelagic environment and enter nearshore developmental habitats such as protected lagoons and open coastal areas rich in sea grass and marine algae. Growth studies using skeletochronology indicate that green sea turtles in the western Atlantic shift from the oceanic phase to nearshore developmental habitats after approximately 5-6 years (Bresette et al. 2006; Zug and Glor 1998). Within the developmental habitats, juveniles begin the switch to a more herbivorous diet, and by adulthood feed almost exclusively on seagrasses and algae (Rebel 1974), although some populations are known to also feed heavily on invertebrates (Carballo et al. 2002). Green sea turtles mature slowly, requiring 20-50 years to reach sexual maturity (Chaloupka and Musick 1997; Hirth 1997).

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

Status and Population Dynamics
Accurate population estimates for marine turtles do not exist because of the difficulty in sampling turtles over their geographic ranges and within their marine environments. Nonetheless, researchers have used nesting data to study trends in reproducing sea turtles over

time. A summary of nesting trends and nester abundance is provided in the most recent status review for the species (Seminoff et al. 2015), with information for each of the DPSs.

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

Quintana Roo, Mexico, accounts for approximately 11% of nesting for the DPS (Seminoff et al. 2015). In the early 1980s, approximately 875 nests/year were deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS 2007d). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpublished data, 2013, in Seminoff et al. 2015).

Tortuguero, Costa Rica is by far the predominant nesting site, accounting for an estimated 79% of nesting for the DPS (Seminoff et al. 2015). Nesting at Tortuguero appears to have been increasing since the 1970's, when monitoring began. For instance, from 1971-1975 there were approximately 41,250 average annual emergences documented and this number increased to an average of 72,200 emergences from 1992-1996 (Bjorndal et al. 1999). Troëng and Rankin (2005) collected nest counts from 1999-2003 and also reported increasing trends in the population consistent with the earlier studies, with nest count data suggesting 17,402-37,290 nesting females per year (NMFS and USFWS 2007). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually. However, a recent long-term study spanning over 50 years of nesting at Tortuguero found that while nest numbers increased steadily over 37 years from 1971-2008, the rate of increase slowed gradually from 2000-2008. After 2008 the nesting trend has been downwards, with current nesting levels having reverted to that of the mid 1990's and the overall long-term trend has now become negative (Restrepo, et al. 2023).

In the continental United States, green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida (Meylan et al. 1994; Weishampel et al. 2003). Occasional nesting has also been documented along the Gulf Coast of Florida (Meylan et al. 1995). Green sea turtle nesting is documented annually on beaches of North Carolina, South Carolina, and Georgia, though nesting is found in low quantities (up to tens of nests) (nesting databases maintained on www.seaturtle.org).

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years. In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 3). According to data collected from Florida's index nesting beach survey from 1989-2021, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 40,911 in 2019. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011. The pattern departed from the low lows and high peaks in 2020 and 2021 as well, when 2020 nesting only dropped by half from the 2019 high, while 2021 nesting only increased by a small amount over the 2020 nesting, with another increase in 2022 still well below the 2019 high (Figure 3). While nesting in Florida has shown dramatic increases over the past decade, individuals from the Tortuguero, the Florida, and the other Caribbean and Gulf of Mexico populations in the North Atlantic DPS intermix and share developmental habitat. Therefore, threats that have affected the Tortuguero population as described previously, may ultimately influence the other population trajectories, including Florida. Given the large size of the Tortuguero nesting population, which is currently in decline, its status and trend largely drives the status of North Atlantic DPS.

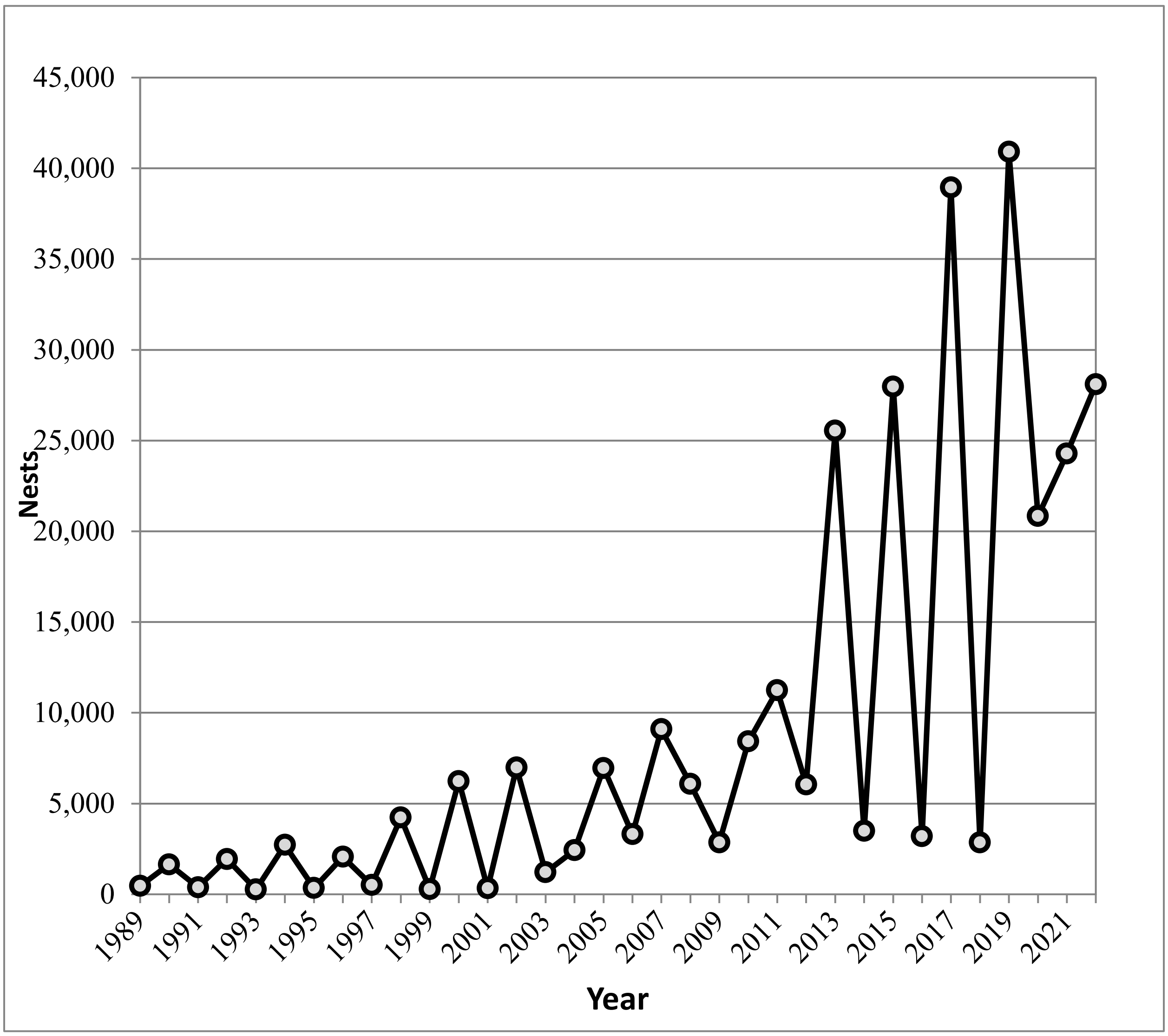

Figure 3. Green sea turtle nesting at Florida index beaches since 1989

Similar to the nesting trend found in Florida, in-water studies in Florida have also recorded increases in green turtle captures at the Indian River Lagoon site, with a 661 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).

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.1.1.

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

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

Whereas oil spill impacts are discussed generally for all species in Section 4.1.1, specific impacts of the DWH spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juveniles greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (DWH Trustees 2015). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil or dispersants, and loss of foraging resources, which could lead to compromised growth and reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

While green turtles regularly use the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic, and the proportion of the population using the northern Gulf of Mexico at any given time is relatively low. Although it is known that adverse impacts occurred and numbers of animals in the Gulf of Mexico were reduced as a result of the Deepwater Horizon oil spill of 2010 (DWH), the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event, as well as the impacts being primarily to smaller juveniles (lower reproductive value than adults and large juveniles), reduces the impact to the overall population. It is unclear what impact these losses may have caused on a population level, but it is not expected to have had a large impact on the population trajectory moving forward. However, recovery of green turtle numbers equivalent to what was lost in the northern Gulf of Mexico as a result of the spill will likely take decades of sustained efforts to reduce the existing threats and enhance survivorship of multiple life stages (DWH Trustees 2015).

4.1.3 Status of Kemp's Ridley Sea Turtle

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

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

each bridge adjoining the plastron to the carapace, there are 4 scutes, each of which is perforated by a pore.

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

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

Life History Information

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

The average rates of growth may vary by location, but generally fall within 2.2-2.9 ± 2.4 in per year (5.5-7.5 ± 6.2 cm/year) (Schmid and Barichivich 2006; Schmid and Woodhead 2000). Age to sexual maturity ranges greatly from 5-16 years, though NMFS et al. (2011) determined the best estimate of age to maturity for Kemp's ridley sea turtles was 12 years. It is unlikely that most adults grow very much after maturity. While some sea turtles nest annually, the weighted mean remigration rate for Kemp's ridley sea turtles is approximately 2 years. Nesting generally occurs from April to July. Females lay approximately 2.5 nests per season with each nest containing approximately 100 eggs (Márquez M. 1994).

Population Dynamics

Of the 7 species of sea turtles in the world, the Kemp's ridley has declined to the lowest population level. Most of the population of adult females nest on the beaches of Rancho Nuevo, Mexico (Pritchard 1969). When nesting aggregations at Rancho Nuevo were discovered in 1947, adult female populations were estimated to be in excess of 40,000 individuals (Hildebrand 1963). By the mid-1980s, however, nesting numbers from Rancho Nuevo and adjacent Mexican beaches were below 1,000, with a low of 702 nests in 1985. Yet, nesting steadily increased through the 1990s, and then accelerated during the first decade of the twenty-first century (Figure 4), which indicated the species was recovering.

It is worth noting that when the Bi-National Kemp's Ridley Sea Turtle Population Restoration Project was initiated in 1978, only Rancho Nuevo nests were recorded. In 1988, nesting data from southern beaches at Playa Dos and Barra del Tordo were added. In 1989, data from the northern beaches of Barra Ostionales and Tepehuajes were added, and most recently in 1996, data from La Pesca and Altamira beaches were recorded. Currently, nesting at Rancho Nuevo accounts for just over 81% of all recorded Kemp's ridley nests in Mexico. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley nests in Mexico increased to 21,797 in 2012 (Gladys Porter Zoo 2013). From 2013 through 2014, there was a second significant decline, as only 16,385 and 11,279 nests were recorded, respectively. More recent data, however, indicated an increase in nesting. In 2015 there were 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Gladys Porter Zoo 2016). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm., August 31, 2017), but nesting for 2018 declined to 17,945, with another steep drop to 11,090 nests in 2019 (Gladys Porter Zoo data, 2019). Nesting numbers rebounded in 2020 (18,068 nests), 2021 (17,671 nests), and 2022 (17,418) (CONAMP data, 2022). At this time, it is unclear whether the increases and declines in nesting seen over the past decade-and-a-half represents a population oscillating around an equilibrium point, if the recent three years (2020-2022) of relatively steady nesting indicates that equilibrium point, or if nesting will decline or increase in the future. So at this point we can only conclude that the population has dramatically rebounded from the lows seen in the 80's and 90's, but we cannot ascertain a current population trend or trajectory.

A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (National Park Service data). It is worth noting that nesting in Texas has somewhat paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015, the record nesting in 2017, and then a drop back down to 190 nests in 2019, rebounding to 262 nests in 2020, back to 195 nests in 2021, and then rebounding to 284 nests in 2022 (National Park Service data).

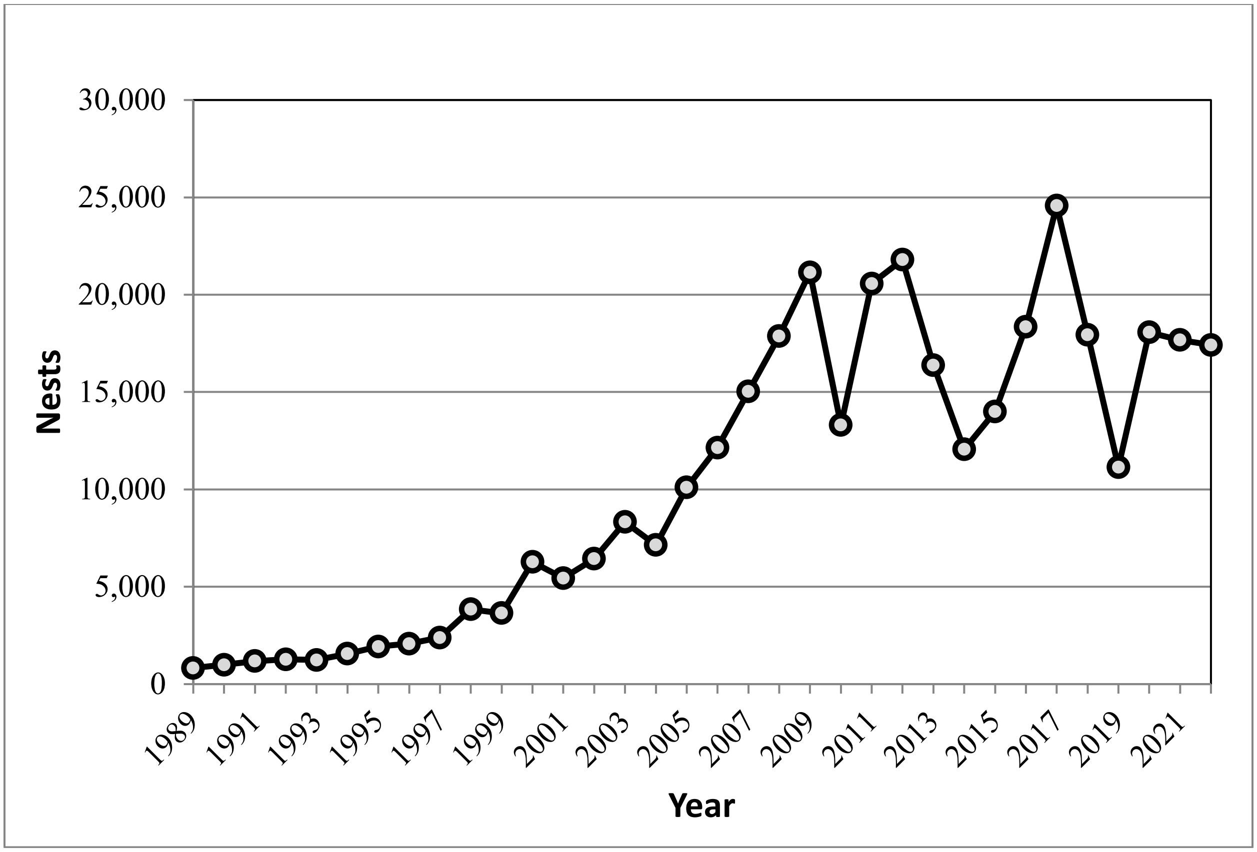

Figure 4. Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019 and CONAMP data 2020-2022)

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 are likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of TEDs, reduced trawling effort in Mexico and the United States, and possibly other changes in vital rates (TEWG 1998; TEWG 2000). While these results are encouraging, the species' limited range as well as low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental randomness, all factors which are often difficult to predict with any certainty. Additionally, the significant nesting declines observed in 2010 and 2013-2014 potentially indicate a serious population-level impact, and the ongoing recovery trajectory is unclear.

Threats

Kemp's ridley sea turtles face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.1.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact Kemp's ridley sea turtles.

As Kemp's ridley sea turtles continue to recover and nesting arribadas (massive, synchronized nesting events) 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 STSSN data, https://www.fisheries.noaa.gov/national/marine-life-distress/sea-turtle-stranding-and-salvage-network) elevated sea turtle strandings in the Northern Gulf of Mexico, particularly throughout the Mississippi Sound area. For example, in the first 3 weeks of June 2010, over 120 sea turtle strandings were reported from Mississippi and Alabama waters, none of which exhibited any signs of external oiling to indicate effects associated with the DWH oil spill event. A total of 644 sea turtle strandings were reported in 2010 from Louisiana, Mississippi, and Alabama waters, 561 (87%) of which were Kemp's ridley sea turtles. During March through May of 2011, 267 sea turtle strandings were reported from Mississippi and Alabama waters alone. A total of 525 sea turtle strandings were reported in 2011 from Louisiana, Mississippi, and Alabama waters, with the majority (455) having occurred from March through July, 390 (86%) of which were Kemp's ridley sea turtles. During 2012, a total of 384 sea turtles were reported from Louisiana, Mississippi, and Alabama waters. Of these reported strandings, 343 (89%) were Kemp's ridley sea turtles. During 2014, a total of 285 sea turtles were reported from Louisiana, Mississippi, and Alabama waters, though the data is incomplete. Of these reported strandings, 229 (80%) were Kemp's ridley sea turtles. These stranding numbers are significantly greater than reported in past years; Louisiana, Mississippi, and Alabama waters reported 42 and 73 sea turtle strandings for 2008 and 2009, respectively. In subsequent years stranding levels during the March-May time period have been elevated but have not reached the high levels seen in the early 2010's. It should be noted that stranding coverage has increased considerably due to the DWH oil spill event.

Nonetheless, considering that strandings typically represent only a small fraction of actual mortality, these stranding events potentially represent a serious impact to the recovery and survival of the local sea turtle populations. While a definitive cause for these strandings has not been identified, necropsy results indicate a significant number of stranded turtles from these events likely perished due to forced submergence, which is commonly associated with fishery interactions (B. Stacy, NMFS, pers. comm. to M. Barnette, NMFS PRD, March 2012). Yet, available information indicates fishery effort was extremely limited during the stranding events. The fact that 80% or more of all Louisiana, Mississippi, and Alabama stranded sea turtles in the past 5 years were Kemp's ridley 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 ridley (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) CCL. Subsequent years of observation noted additional captures in the skimmer trawl fisheries, including some mortalities. The small average size of encountered Kemp's ridley sea turtles introduces a potential conservation issue, as over 50% of these reported sea turtles could potentially pass through the maximum 4-in bar spacing of TEDs currently required in the shrimp fisheries. Due to this issue, a proposed 2012 rule to require 4-in bar spacing TEDs in the skimmer trawl fisheries (77 FR 27411) was not implemented. Following additional gear testing, however, we proposed a new rule in 2016 (81 FR 91097) to require TEDs with 3-inch (in) bar spacing for all vessels using skimmer trawls, pusher-head trawls, or wing nets. Ultimately, we published a final rule on December 20, 2019 (84 FR 70048), that requires all skimmer trawl vessels 40 feet and greater in length to use TEDs designed to exclude small sea turtles in their nets effective April 1, 2021. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of Mexico may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 4.1.1, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles are considered here. Kemp's ridley sea turtles 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 ridley 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 ridley sea turtles (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 ridley 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 Kemp's ridley juveniles are estimated to have died as a direct result of the exposure. Therefore, as much as 20% of the small oceanic juveniles of this species were killed during that year. Impacts to large juveniles (>3 years old) and adults were also high. An estimated 21,990 such individuals were exposed to oil (about 22% of the total estimated population for those age classes); of those, 3,110 mortalities were estimated (or 3% of the population for those age classes). The loss of near-reproductive and reproductive-stage females would have contributed to some extent to the decline in total nesting abundance observed between 2011 and 2014. The estimated number of unrealized Kemp's ridley nests is between 1,300 and 2,000, which translates to between approximately 65,000 and 95,000 unrealized hatchlings (DWH Trustees 2016). This is a minimum estimate, however, because the sublethal effects of the DWH oil spill event on turtles, their prey, and their habitats might have delayed or reduced reproduction in subsequent years, which may have contributed substantially to additional nesting deficits observed following the DWH oil spill event. These sublethal effects could have slowed growth and maturation rates, increased remigration intervals, and decreased clutch frequency (number of nests per female per nesting season). The nature of the DWH oil spill event effect on reduced Kemp's ridley nesting abundance and associated hatchling production after 2010 requires further evaluation. It is clear that the DWH oil spill event resulted in large losses to the Kemp's ridley population across various age classes, and likely had an important population-level effect on the species. Still, we do not have a clear understanding of those impacts on the population trajectory for the species into the future.

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

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

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

Life History Information

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

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

Female leatherbacks typically nest on sandy, tropical beaches at intervals of 2-4 years (Garcia M. and Sarti 2000; McDonald and Dutton 1996; Spotila et al. 2000). Unlike other sea turtle species, female leatherbacks do not always nest at the same beach year after year; some females may even nest at different beaches during the same year (Dutton et al. 2005; Eckert 1989; Keinath and Musick 1993; Steyermark et al. 1996). Individual female leatherbacks have been observed with fertility spans as long as 25 years (Hughes 1996). Females usually lay up to 10 nests during the 3-6 month nesting season (March through July in the United States), typically 8-12 days apart, with 100 eggs or more per nest (Eckert et al. 2012; Eckert 1989; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). Yet, up to approximately 30% of the eggs may be infertile (Eckert 1989; Eckert et al. 1984; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50% worldwide (Eckert et al. 2012), which is lower than the greater than 80% reported for other sea turtle species (Miller 1997). In the United States, the emergent success is higher at 54-72% (Eckert and Eckert 1990; Stewart and Johnson 2006; Tucker 1988). Thus the number of hatchlings in a given year may be less than the total number of eggs produced in a season. Eggs hatch after 60-65 days, and the hatchlings have white striping along the ridges of their backs and on the edges of the flippers. Leatherback hatchlings weigh approximately 1.5-2 oz (40-50 g), and have lengths of approximately 2-3 in (51-76 mm), with fore flippers as long as their bodies. Hatchlings grow rapidly, with reported growth rates for leatherbacks from 2.5-27.6 in (6-70 cm) in length, estimated at 12.6 in (32 cm) per year (Jones et al. 2011).

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

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

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

Status and Population Dynamics

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

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

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

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. Across the Western Caribbean, nesting is most prevalent in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coastline of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from index nesting beaches in Tortuguero, Gandoca, and Pacuaré in Costa Rica indicate that the nesting population likely was not growing over the 1995-2005 time series (TEWG 2007). Other modeling of the nesting data for Tortuguero indicates a possible 67.8% decline between 1995 and 2006 (Troëng et al. 2007). Tiwari et al. (2013) report an estimated three-generation abundance change of -72%, -24%, and +6% for Tortuguero, Gandoca, and Pacuare, respectively. Further decline of almost 6% annual geometric mean from 2008-2017 reflects declines in nesting beaches throughout this stock (Northwest Atlantic Leatherback Working Group 2018).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, St. Croix (U.S. Virgin Islands), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (TEWG 2007). Tiwari et al. (2013) report an estimated three-generation abundance change of -4% and +5,583% at Culebra and Fajardo, respectively. At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has varied from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (TEWG 2007). From 2006-2010, Tiwari et al. (2013) report an annual growth rate of +7.5% in St. Croix and a three-generation abundance change of +1,058%. Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2% between 1994 and 2004 (TEWG 2007). The nesting trend reversed course later, with an annual geometric mean decline of 10% from 2008-2017 driving the long-term trend (1990-2017) down to a 2% annual decline (Northwest Atlantic Leatherback Working Group 2018).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. FWC Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 5 and Table 3). A similar pattern was also observed statewide (Table 3). 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, nesting declined dramatically on Florida beaches from 2014-2017, with 2017 hitting a decade-low number. The decline was then followed by a partial rebound from 2018 to 2022. The annual geometric mean trend for Florida had been a decline of almost 7% from 2008-2017, but the long-term trend (1990-2017) remains positive with an annual geometric mean increase of over 9% (Northwest Atlantic Leatherback Working Group 2018). The increase since the 2017 low further solidifies the long-term increase, while the pattern in the past decade is less certain.

Table 3. Number of Leatherback Sea Turtle Nests in Florida

Leatherback Nests Recorded- Florida		
Year	**Index Nesting Beach Survey**	**Statewide Survey**
2011	625	1,653
2012	515	1,712
2013	322	896
2014	641	1,604
2015	489	1,493
2016	319	1,054
2017	205	663
2018	316	949
2019	337	1,105
2020	467	1,652
2021	435	1,390
2022	514	1,848

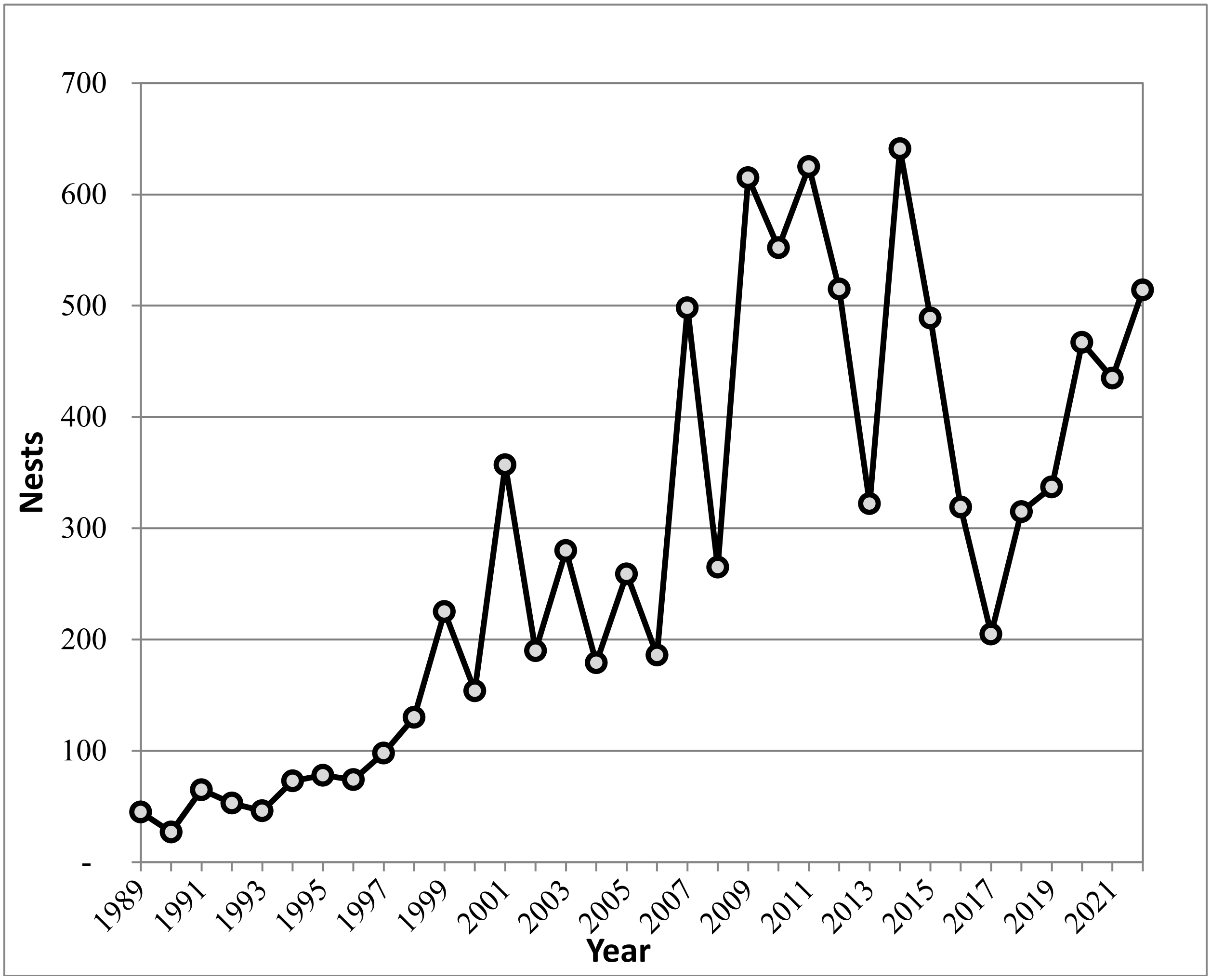

Figure 5. Leatherback sea turtle nesting at Florida index beaches since 1989

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

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

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

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

Threats

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.2.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

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

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

As discussed in Section 4.1.1, 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 2007). Several studies have shown leatherback distribution is influenced by jellyfish abundance (Houghton et al. 2006; Witt et al. 2007; Witt et al. 2006); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 4.1.1, 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. 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 or dispersants, and loss of foraging resources which could lead to compromised growth and 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.

4.1.5 Status of Loggerhead Sea Turtle – Northwest Atlantic DPS

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

Species Description and Distribution

Loggerheads are large sea turtles. Adults in the southeast 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 vertebrals, and a nuchal (precentral) scute that is in contact with the first pair of costal scutes (Dodd Jr. 1988).

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

The majority of loggerhead nesting occurs at the western rims of the Atlantic and Indian Oceans concentrated in the north and south temperate zones and subtropics (NRC 1990). For the Northwest Atlantic DPS, most nesting occurs along the coast of the United States, from southern Virginia to Alabama. Additional nesting beaches for this DPS are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba (Gavilan 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

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

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

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles concluded that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida Peninsula. It also concluded that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the recovery plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are as follows: (1) the Northern Recovery Unit (Florida/Georgia border north through southern Virginia), (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The recovery plan concluded that all recovery units are essential to the recovery of the species. Although the recovery plan was written prior to the listing of the Northwest Atlantic DPS, the recovery units for what was then termed the Northwest Atlantic population apply to the Northwest Atlantic 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-nearshore marine environment from the surface to the sea floor where water depths do not exceed 200 meters), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting female (terrestrial zone) (NMFS and USFWS 2008). Loggerheads are long-lived animals. They reach sexual maturity between 20-38 years of age, although age of maturity varies widely among populations (Frazer and Ehrhart 1985; NMFS 2001). The annual mating season occurs from late March to early June, and female turtles lay eggs throughout the summer months. Females deposit an average of 4.1 nests within a nesting season (Murphy and Hopkins 1984), but an individual female only nests every 3.7 years on average (Tucker 2010). Each nest contains an average of 100-126 eggs (Dodd Jr. 1988) which incubate for 42-75 days before hatching (NMFS and USFWS 2008). Loggerhead hatchlings are 1.5-2 inches long and weigh about 0.7 oz (20 g).

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

After departing the oceanic zone, neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, the Bahamas, Cuba, and the Gulf of Mexico. Estuarine waters of the United States, including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River

Lagoons, Biscayne Bay, Florida Bay, as well as numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads (Conant et al. 2009).

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

Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has also been documented (Hawkes et al. 2007) GADNR, unpublished data; SCDNR, unpublished data). Satellite telemetry has identified the shelf waters along the west Florida coast, the Bahamas, Cuba, and the Yucatán Peninsula as important resident areas for adult female loggerheads that nest in Florida (Foley et al. 2008; Girard et al. 2009; Hart et al. 2012). The southern edge of the Grand Bahama Bank is important habitat for loggerheads nesting on the Cay Sal Bank in the Bahamas, but nesting females are also resident in the bights of Eleuthera, Long Island, and Ragged Islands. They also reside in Florida Bay in the United States, and along the north coast of Cuba (A. Bolten and K. Bjorndal, University of Florida, unpublished data). Moncada et al. (2010) report the recapture of 5 adult female loggerheads in Cuban waters originally flipper-tagged in Quintana Roo, Mexico, which indicates that Cuban shelf waters likely also provide foraging habitat for adult females that nest in Mexico.

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

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

<u>Peninsular Florida Recovery Unit</u>
The PFRU is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to

2007 showed an average of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (NMFS and USFWS 2008). The statewide estimated total for 2020 was 105,164 nests (FWRI nesting database).

In addition to the total nest count estimates, the Florida Fish and Wildlife Research Institute (FWRI) uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. FWRI uses the standardized index survey data to analyze the nesting trends (Figure 6) (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). Since the beginning of the index program in 1989, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that was followed by a sharp decline over the subsequent 9 years. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represented a new record for loggerheads on the core index beaches. While nest numbers subsequently declined from the 2016 high FWRI noted that the 2007-2021 period represents a period of increase. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but not significant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening confidence intervals. Nesting at the core index beaches declined in 2017 to 48,033, and rose again each year through 2020, reaching 53,443 nests, dipping back to 49,100 in 2021, and then in 2022 reaching the second-highest number since the survey began, with 62,396 nests. It is important to note that with the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.) it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani, et al. 2019).

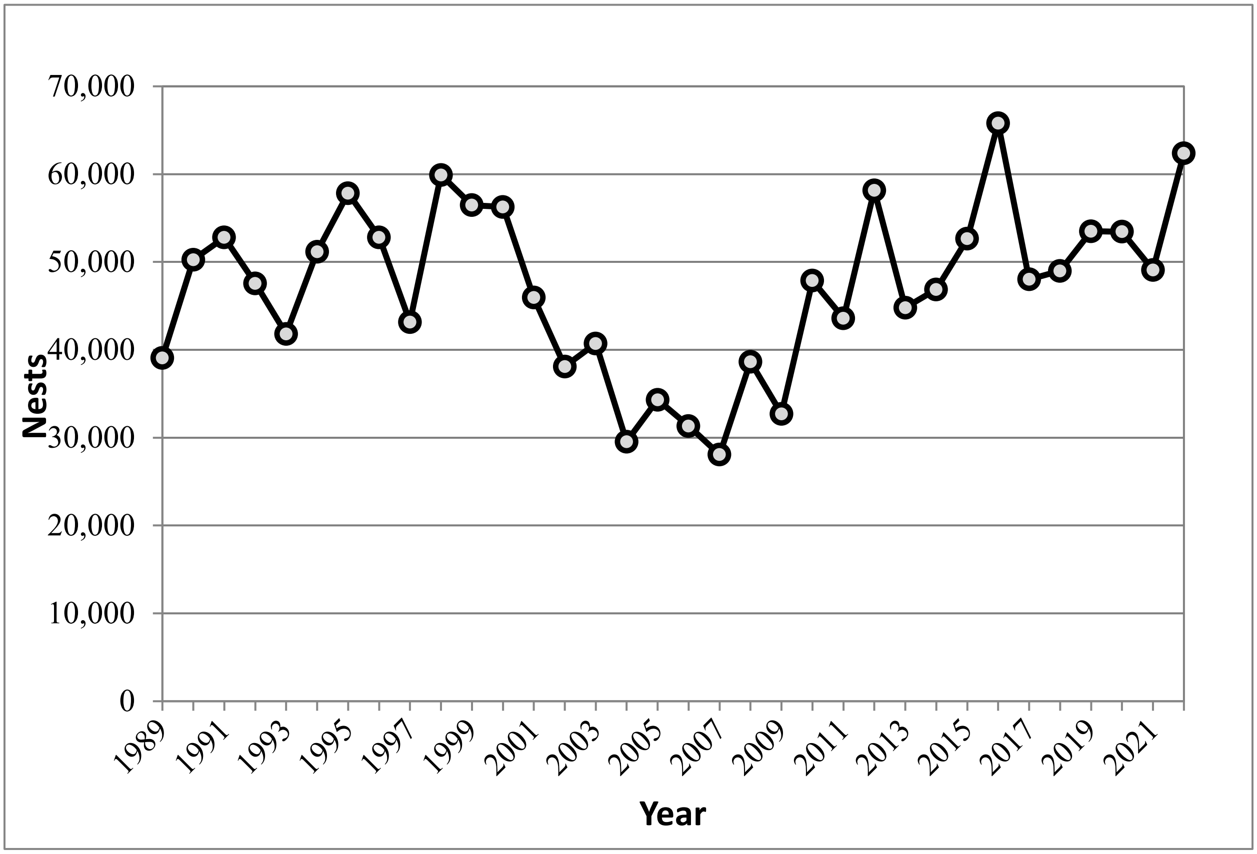

Figure 6. Loggerhead sea turtle nesting at Florida index beaches since 1989

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

Data since that analysis (Table 4) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, GADNR press release, https://georgiawildlife.com/loggerhead-nest-season-begins-where-monitoring-began). South Carolina and North Carolina nesting have also begun to shift away from the past declining trend. Loggerhead nesting in Georgia, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016. Nesting in 2017 and 2018 declined relative to 2016, back to levels seen in 2013 to 2015, but then bounced back in 2019, breaking records for each of the three states and the overall recovery unit. Nesting in 2020 and 2021 declined from the 2019 records, but still remained high, representing the third and fourth highest total numbers for the NRU since 2008. In 2022 Georgia loggerhead nesting broke the record at 4,071, while South Carolina and North Carolina nesting were both at the second-highest level recorded.

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

Year	Georgia	South Carolina	North Carolina	Totals
2008	1,649	4,500	841	**6,990**
2009	998	2,182	302	**3,472**
2010	1,760	3,141	856	**5,757**
2011	1,992	4,015	950	**6,957**
2012	2,241	4,615	1,074	**7,930**
2013	2,289	5,193	1,260	**8,742**
2014	1,196	2,083	542	**3,821**
2015	2,319	5,104	1,254	**8,677**
2016	3,265	6,443	1,612	**11,320**
2017	2,155	5,232	1,195	**8,582**
2018	1,735	2,762	765	**5,262**
2019	3,945	8,774	2,291	**15,010**
2020	2,786	5,551	1,335	**9,672**
2021	2,493	5,639	1,448	**9,580**
2022	4,071	7,970	1,906	**13,947**

In addition to the statewide nest counts, South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record. After another drop in 2018, a new record was set for the 2019 season, with a return to 2016 levels in 2020 and 2021 and then a rebound to the second highest level on record in 2022 (Figure 7).

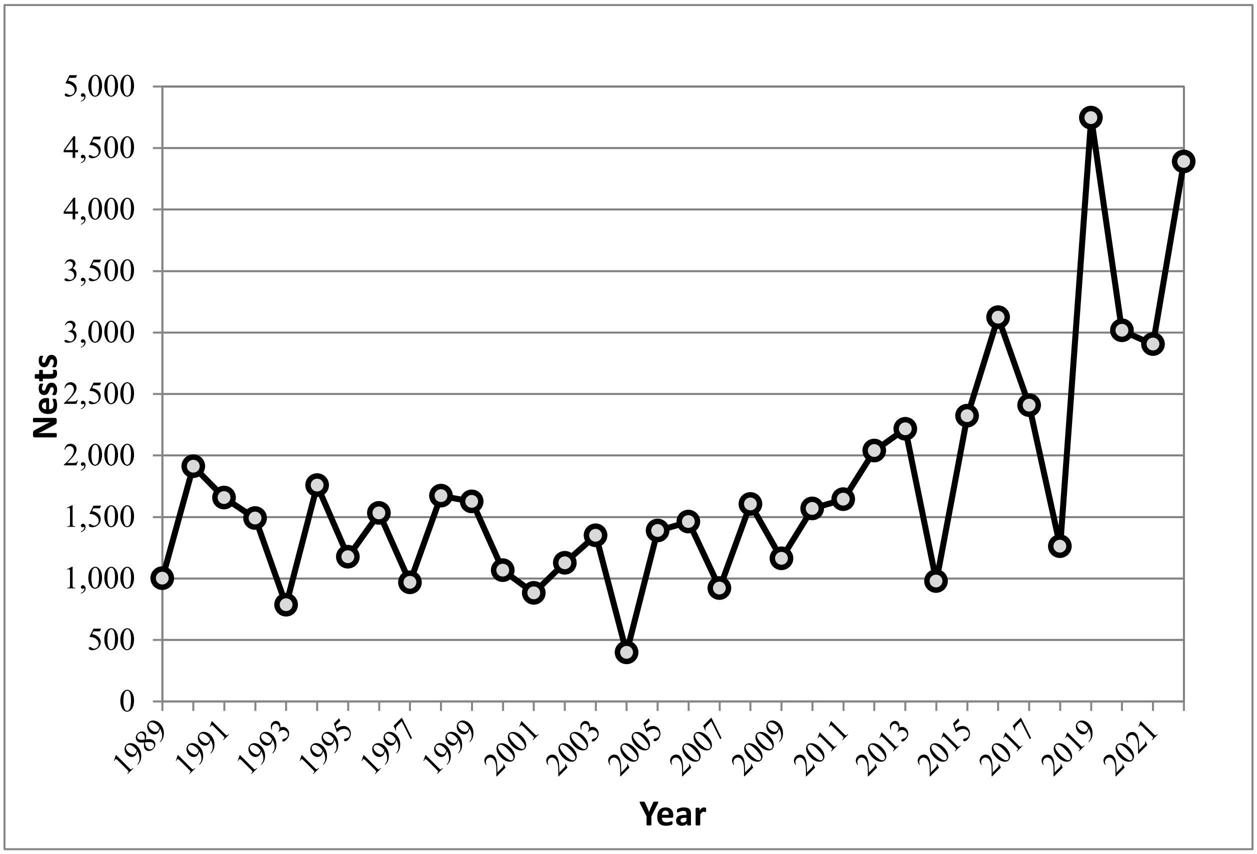

Figure 7. South Carolina index nesting beach counts for loggerhead sea turtles (data provided by SCDNR)

<u>Other Northwest Atlantic DPS Recovery Units</u>

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. From 1989-2018 the average number of NGMRU nests annually on index beaches was 169 nests, with an average of 1100 counted in the statewide nesting counts (Ceriani et al. 2019). Nesting survey effort has been inconsistent among the GCRU nesting beaches, and no trend can be determined for this subpopulation (NMFS and USFWS 2008). Zurita et al. (2003) found a statistically significant increase in the number of nests on 7 of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. Nonetheless, nesting has declined since

2001, and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008).

In-water Trends

Nesting data are the best current indicator of sea turtle population trends, but in-water data also provide some insight. In-water research suggests the abundance of neritic juvenile loggerheads is steady or increasing. Although Ehrhart et al. (2007) found no significant regression-line trend in a long-term dataset, researchers have observed notable increases in CPUE (Arendt et al. 2009; Ehrhart et al. 2007; Epperly et al. 2007). Researchers believe that this increase in CPUE is likely linked to an increase in juvenile abundance, although it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. Bjorndal et al. (2005), cited in NMFS and USFWS (2008), caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern United States may be due to increased abundance of the largest oceanic/neritic juveniles (historically referred to as small benthic juveniles), which could indicate a relatively large number of individuals around the same age may mature in the near future (TEWG 2009). In-water studies throughout the eastern United States, however, indicate a substantial decrease in the abundance of the smallest oceanic/neritic juvenile loggerheads, a pattern corroborated by stranding data (TEWG 2009).

Population Estimate

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

Threats (Specific to Loggerhead Sea Turtles)

The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 4.1.1. 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 Northwest Atlantic 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. Dietary preferences were likely to be the main differentiating factor among sea turtle species. Storelli et al. (2008) analyzed tissues from stranded loggerhead sea turtles and found that mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991).

While oil spill impacts are discussed generally for all species in Section 4.1.1, specific impacts of the DWH oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (DWH Trustees 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil or dispersants, and loss of foraging resources which could lead to compromised growth and reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

Unlike Kemp's ridleys, the majority of nesting for the Northwest Atlantic 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 Northwest Atlantic DPS would be proportionally much greater than the impacts occurring to other recovery units. Impacts to nesting and oiling effects on a large proportion of the NGMRU recovery unit, especially mating and nesting adults likely had an impact on the NGMRU. Based on the response injury evaluations for Florida Panhandle and Alabama nesting beaches (which fall under the NFMRU), the DWH Trustees (2016) estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH oil spill response activities on nesting beaches. Although the long-term effects remain unknown, the DWH oil spill event impacts to the NGMRU 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).

5 ENVIRONMENTAL BASELINE

5.1 Overview

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

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

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

5.2 Baseline Status of ESA-Listed Species Considered for Further Analysis

As stated in Section 2.2 (Action Area), the proposed action occurs in the Gulf of Mexico, approximately 20.9 nm offshore Mexico Beach, Bay County, Florida. As discussed in Section 3.1, four species of ESA-listed sea turtles may be adversely affected by the proposed action. These species are all highly migratory. The status of these species in the action area, as well as the threats to these species, are the same as those discussed in Section 4 (Status of the Species).

5.3 Additional Factors Affecting the Baseline Status of ESA-Listed Species Considered for Further Analysis

5.3.1 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, have 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 area which have already concluded or are currently undergoing formal Section 7 consultation.

5.3.1.1 Federal Vessel Activity

Watercraft are the greatest contributors to overall noise in the sea and have the potential to interact with sea turtles though direct impacts or propellers. Sound levels and tones produced are generally related to vessel size and speed. Larger vessels generally emit more sound than smaller vessels, and vessels underway with a full load, or those pushing or towing a load, are noisier than unladen vessels. Vessels operating at high speeds have the potential to strike sea turtles. Potential sources of adverse effects from federal vessel operations in the action area include operations of the BOEM, FERC, USCG, NOAA, BSEE, U.S. EPA, USFWS, and USACE.

We have conducted Section 7 consultations related to energy projects in the Gulf of Mexico (BOEM, FERC, BSEE, U.S. EPA, 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, USFWS) 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 or operational activities that are unlikely to contribute a large amount of risk.

5.3.1.2 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 mi 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.

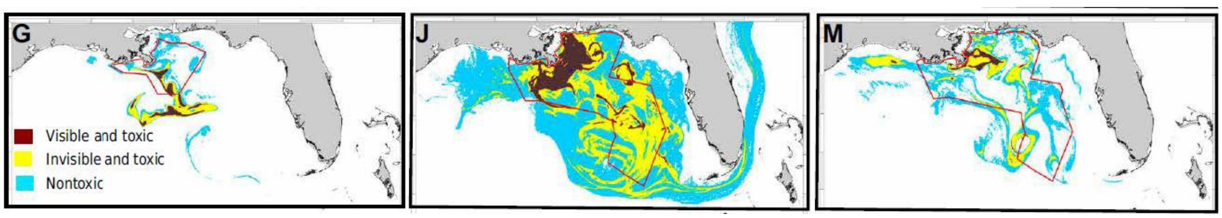

Figure 8. The spread of the impacts from the DWH spill; G from 15 May 2010, J from 18 June 2010, M from 2 July 2010 (Berenshtein et al. 2020).

5.3.1.3 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.

5.3.1.4 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.

5.3.1.5 Department of Defense Training Activities

The Eglin Gulf Test and Training Range comprises 102,000 nm^2 of Gulf of Mexico surface waters, beginning 3 nm from shore. NMFS previously consulted on maritime strike missions in the Gulf of Mexico involving the use of multiple types of live munitions against small boat targets in the Eglin Gulf Test and Training Range (SER-2012-9587). The Opinion included an ITS and determined that strike operations would not jeopardize any species of green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), or leatherback sea turtle or other ESA-listed species, and would not destroy or adversely modify critical habitat of any listed species.

5.3.2 State and Private Actions

5.3.2.1 State Fisheries

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

Trawl Fisheries
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.

5.3.2.2 Recreational Fishing

Recreational fishing as regulated by Florida 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).

5.3.2.3 Artificial Reefs

Bay County has a very active artificial reef program. There are numerous artificial reef and shipwreck sites located in close proximity to the proposed action area (Figure 8). Reef structures range from reef modules (green triangles) to concrete rubble (beige pyramids), and from metal (red triangles) to sunken vessels (ship icon). Impacts of artificial reefs on sea turtles are described in both the Effects of Action (Section 5) below and in Barnette (2017).

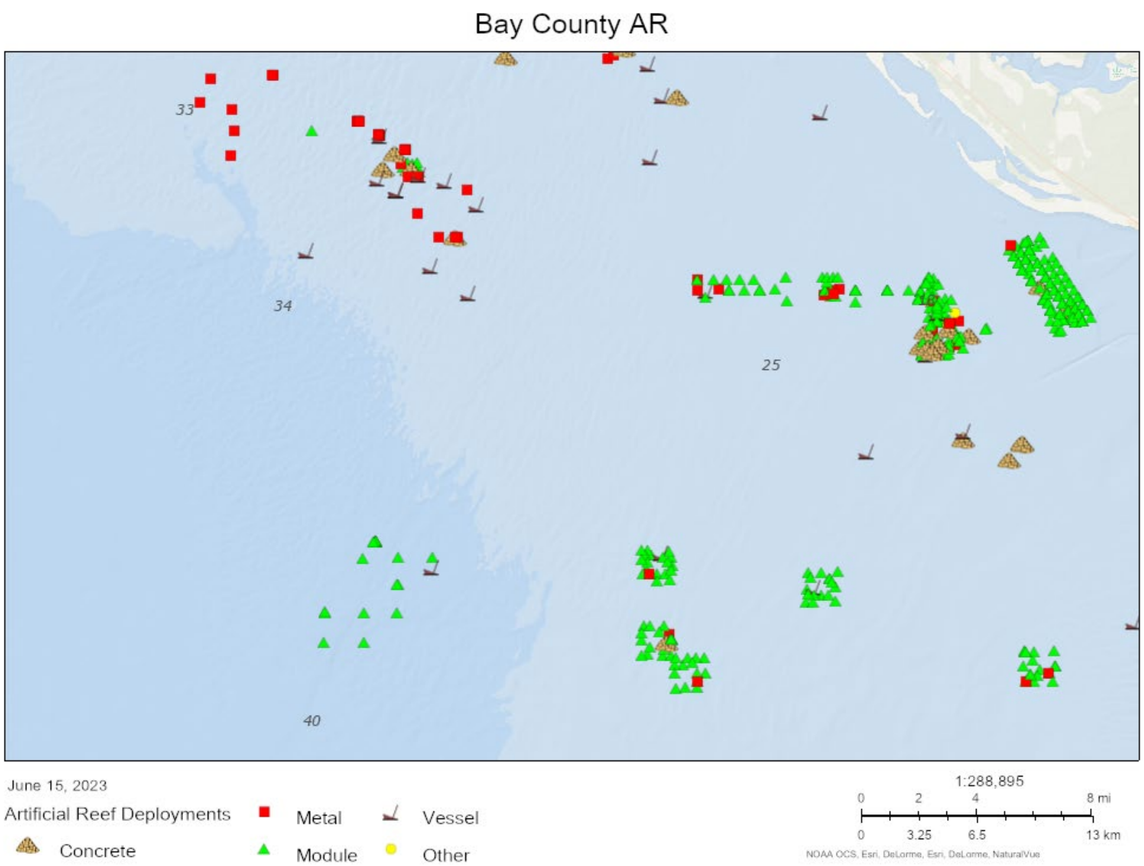

Figure 9. Locations of Existing Artificial Reef Structures in and around Bay County, Florida

(**https://myfwc.maps.arcgis.com/apps/View/index.html?appid=4675e1db32ac43a9a4308e757965d17d**).

5.3.2.4 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 Zone 8 (which includes the action area) includes 56 records of vessel interactions with sea turtles, of which all but 3 were fatal. 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 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.

5.3.2.5 Coastal Development

Beachfront development, lighting, and beach erosion control all are ongoing activities along the Florida coastline, including the barrier islands near the action area. ESA consultations within or near the action area include dredging and beach nourishment along the shoreline of Bay County (SERO-2022-02839, Mexico Beach Dune and Beach Restoration; issued June 14, 2023). 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.

5.3.3 Marine Debris, 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.

5.3.4 Stochastic Events

Stochastic 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.

5.3.5 Climate Change

As discussed in Section 4.1.1 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). A shift to higher temperatures could also affect hatchling sex ratios resulting in a higher number of females. 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.

6 EFFECTS OF THE ACTION

6.1 Overview

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 the effect would not occur but for the proposed action and the effect is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (50 CFR 402.02).

In this section of our Opinion, we assess the effects of the action on listed species that are likely to be adversely affected. The analysis in this section forms the foundation for our jeopardy analysis in Section 8. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Data are limited, so we 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. NMFS generally selects the value that would lead to conclusions of higher, rather than lower risk to endangered or threatened species. This approach provides the “benefit of the doubt” to threatened and endangered species.

6.2 Effects of the Proposed Action on ESA-Listed Species Considered for Further Analysis

6.2.1 Routes of Effect That Are Not Likely to Adversely Affect ESA-Listed Species

Routes of effect that are not likely to adversely affect ESA-listed sea turtles are discussed in Section 3.1.2.

6.2.2 Routes of Effect That Are Likely to Adversely Affect ESA-Listed Species

NMFS believes that the presence of high-relief artificial reef material is likely to adversely affect the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS). 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^2 (individually or collectively), excluding prefabricated artificial reef modules.

Because artificial reefs are generally designed and advertised to promote fishing opportunities, sea turtles may be adversely affected by becoming entangled in lost fishing gear and marine debris that accumulates on these structures (e.g., discarded fishing line, anchor line, or discarded netting). The risk of entanglement increases over the lifespan of the artificial reef structure as more gear and debris accumulates (Barnette 2017). Our assessment of this risk and its effects on sea turtles are discussed in more detail below.

Approach to Assessment
Our analysis first reviews what activities associated with the proposed action are likely to adversely affect sea turtles in the action area (i.e., what the stressors of the proposed action are). We then review an individual's range of responses to a specific stressor, and the factors affecting the likelihood, frequency, and severity of an individual's exposure to that stressor. Subsequently, 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 the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS) from entanglement and drowning in monofilament and other entangling gear that accumulates on that type of reef material. Given the complex habitat and vertical relief afforded by these materials, it is not uncommon for these sites to accumulate a significant amount of lost fishing gear over time (Barnette 2017).

In general, due to the absence of monofilament immediately following deployment of an artificial reef, 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 the 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 inter-nesting 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 inter-nesting 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). In this case, the proposed reef deployment area in this proposed action is in water depths between 94 and 100 ft.

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 consider all complex, high-relief materials deployed as artificial reefs (excluding prefabricated artificial reef muddles) to present similar entanglement risks to sea turtles over time, regardless of their location.

Barnette (2017) documents that an historic shipwreck submerged for more than 120 years (i.e., foundered in 1897) appears to be still accumulating monofilament and resulting in sea turtle mortalities due to entanglement events. Given the remaining structure on that shipwreck, 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). Barnette (2017) 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 (i.e., 1 sea turtle mortality per year) 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 1 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 structure.

6.2.2.1 Estimating Total Sea Turtle Mortalities

To calculate the overall sea turtle mortalities for the proposed action, we begin with the assumption that the typical lifespan of 1 structure of high-relief artificial reef material (i.e., a vessel, decommissioned oil rig, bridge span, or other large metal structure) is 150 years. Next, we assume deployment of 1 structure of high-relief artificial reef material will result in the following rates of mortality due to entanglement over 150 years: (1) during the first 25 years, we assume there will be 0 sea turtle mortalities; (2) for the next 75 years, there will be 1 sea turtle mortality each year; and (3) for the last 50 years, we assume there will be 1 sea turtle mortality every 3 years.

The proposed project will result in the annual deployment of a maximum of approximately 5 structures per year in the Sandy Reef deployment area. The life of the proposed USACE permit is 10 years, therefore we estimate that there will be up to 50 deployed structures over the life of the proposed action (5 deployments per year x 10 years = 50 deployments per reef site). Below, we calculate the total number of sea turtle mortalities anticipated at the Sandy Reef deployment site. This is likely an overestimate as not every deployment will be high-relief materials, and the estimated number of deployments per year represents a maximum of expected deployments (i.e., 5 structures).

Years 0-25 =	*0 sea turtle mortalities*
Years 26-100 =	*1 sea turtle mortality per year per structure × 75 years = 75 sea turtle mortalities per structure × 50 structures = 3,750 total sea turtle mortalities*
Years 101-150 =	*50 years × (1 sea turtle mortality ÷ 3 years) = 16.667 sea turtle mortalities per structure × 50 structures = 833.35 rounded up for whole organism estimate = 834 sea turtle mortalities*

Total for 150 years = 3,750 + 834 = 4,584 total sea turtle mortalities

In total, the number of sea turtle mortalities over 150 years resulting from the deployment of high-relief artificial reef materials at Sandy Reef is estimated to be 4,584 sea turtles. The annual average number of sea turtle mortalities is 36.67 (i.e., 4,584 sea turtles ÷ 125 years) sea turtles per year per reef site once mature (i.e., 25 years after the last deployment).

6.2.2.2 Estimating Species Take Percentages

We used the 2007-2016 STSSN data for offshore Zone 8, a statistical sub-area used when reporting commercial fishing data, which includes the action area, to determine the expected number of mortalities for each species within the action area. The 10-year dataset for Zone 8 shows a total of 347 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 mortalities by species resulting from the proposed action (Table 5). Although one hawksbill sea turtle is represented in the data, the cause of death is unknown, and the animal exhibited oil/tar contamination. We believe the presence of hawksbill sea turtles within the action area will be rare, and it is extremely unlikely they would be found interacting with artificial reef material.

Table 5. 2007-2016 STSSN Data for Offshore Florida Gulf Zone 8.

Species	Total Strandings 2007-2016	Species Percent Composition
Green	47	13.54
Hawksbill	1	0.29
Kemp's ridley	149	42.94
Leatherback	1	0.29
Loggerhead	149	42.94
Grand Total	**347**	**100**

To calculate the number of expected sea turtle mortalities broken down by species, we use the following equation, results of which are summarized in Table 6, below.

Expected mortalities by species for 1 high-relief artificial reef over a 150-year time frame out of 4,584 anticipated total sea turtle mortalities
*= total expected sea turtle mortalities over 150 years from artificial reefs (4,584) × percent composition from stranding data for each species (*Table 5*)*

Expected number of <u>*green*</u> *sea turtle mortalities over 150 years*
= 4,584 × 0.1354 = 620.67
Expected number of <u>*Kemp's ridley*</u> *sea turtle mortalities over 150 years*
= 4,584 × 0.4294 = 1,968.37
Expected number of <u>*leatherback*</u> *sea turtle mortalities over 150 years*
= 4,584 × 0.0029 = 13.30
Expected number of <u>*loggerhead*</u> *sea turtle mortalities over 150 years*
= 4,584 × 0.4294 = 1,968.37

Table 6. Breakdown of Lethal Sea Turtle Entanglements Based on STSSN Data (2007-2016) by Species.

Species	Percent from Stranding Data	Species Breakdown Out of 4,584 Anticipated Sea Turtle Takes	Take Estimate Rounded Up
Green (North Atlantic DPS)	13.54%	620.67	621
Kemp's ridley	42.94%	1,968.37	1,969
Leatherback	0.29%	13.30	14
Loggerhead (Northwest Atlantic DPS)	42.94%	1,968.37	1,969
Total	**100**	**4,571.71**	**4,573**

Table 7 summarizes the total number of anticipated lethal entanglements over a period of 150 years for each sea turtle species. We took the total number of sea turtle mortalities expected for each time period of reef aging and multiplied it by the species percentages in Table 5 (e.g., 3,750 mortalities in YR 26-100 × 0.4294 Kemp's ridley sea turtles = 1,611 Kemp's ridley sea turtle mortalities during YR 26-100 of the life of the reef). All calculated values are rounded to the nearest whole number. We note excluding the hawksbill sea turtle results in a slightly lower total number of sea turtle takes than those calculated above (i.e., 4,575 instead of 4,584).

Table 7. Anticipated Amount of Sea Turtle Mortalities, by Species, Over 150 Years due to Entanglements Associated with High-Relief Artificial Reef Material in the Sandy Reef Deployment Area.

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
Green sea turtle (North Atlantic DPS)	0	508	113	621

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
Kemp's ridley sea turtle	0	1,611	359	1,970
Leatherback sea turtle	0	11	3	14
Loggerhead sea turtle (Northwest Atlantic DPS)	0	1,611	359	1,970
Total sea turtle mortality by time period	**0**	**3,741**	**834**	**4,575**

7 CUMULATIVE EFFECTS

ESA Section 7 regulations require NMFS to consider cumulative effects in formulating its Opinions (50 CFR 402.14). Cumulative effects include the effects of future state or private actions, not involving federal activities, that are reasonably certain to occur within the action area considered in this Opinion (50 CFR 402.02). NMFS is not aware of any future projects that may contribute to cumulative effects. Within the action area, the ongoing activities and processes described in the environmental baseline are expected to continue and NMFS did not identify any additional sources of potential cumulative effect.

8 JEOPARDY ANALYSIS

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

The NMFS and USFWS's ESA Section 7 Handbook (USFWS and NMFS 1998) defines survival and recovery, as these terms apply to the ESA's jeopardy standard. Survival means "the species' persistence…beyond the conditions leading to its endangerment, with sufficient resilience to allow recovery from endangerment." The Handbook further explains that survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a sufficiently large population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter. Per the Handbook and the ESA regulations at 50 CFR 402.02, recovery means "improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the Act." Recovery is the process by which species' ecosystems are restored or threats to the species are removed or both so that self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

The analyses conducted in the previous sections of this Opinion serve to provide a basis to determine whether the proposed action would be likely to jeopardize the continued existence of green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS). In Section 6, we outlined how the proposed action can adversely affect these species. Now we turn to an assessment of the species response to these impacts, in terms of overall population effects, and whether those effects of the proposed action, when considered in the context of the Status of the Species (Section 4), the Environmental Baseline (Section 5), and the Cumulative Effects (Section 7), will jeopardize the continued existence of the affected species. For any species listed globally, our jeopardy determination must evaluate whether the proposed action will appreciably reduce the likelihood of survival and recovery at the species' global range. For any species listed as DPSs, a jeopardy determination must evaluate whether the proposed action will appreciably reduce the likelihood of survival and recovery of that DPS.

8.1 Green Sea Turtle (North Atlantic DPS)

As discussed in Section 4.1.2, only individuals from the North Atlantic DPS and South Atlantic DPS may occur in waters under the purview of the NMFS Southeast Region, with South Atlantic DPS individuals only expected to occur in the U.S. Caribbean. The action area is located in the Gulf of Mexico, therefore only individuals from the North Atlantic DPS are expected to be present. The proposed action may result in the lethal take of 621 green sea turtles from the North Atlantic DPS over the next 150 years. The take is expected to be no green sea turtles during the first 25 years, 508 during the next 75 years, and 113 during the last 50 years.

Survival

The potential lethal take of up to 621 green sea turtles from the North Atlantic DPS over the next 150 years from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area 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 taken 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 508 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 opportunistically as materials and funding become available and deployments will occur only within a discrete area. Because green sea turtles from the North Atlantic 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. The North Atlantic 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 (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). A recent long-term study spanning over 50 years of nesting at Tortuguero found that while nest numbers increased steadily over 37 years from 1971-2008, the rate of increase slowed gradually from 2000-2008. After 2008, the nesting trend has been downwards, with current nesting levels having reverted to that of the mid-1990's, and the overall long-term trend has now become negative (Restrepo, et al. 2023).

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). According to data collected from Florida's index nesting beach survey from 1989-2021, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 40,911 in 2019. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011. The pattern departed from the low lows and high peaks in 2020 and 2021 as well, when 2020 nesting only dropped by half from the 2019 high, while 2021 nesting only increased by a small amount over the 2020 nesting, with another increase in 2022 still well below the 2019 high. While nesting in Florida has shown dramatic increases over the past decade, individuals from the Tortuguero, the Florida, and the other Caribbean and Gulf of Mexico populations in the North Atlantic DPS intermix and share developmental habitat. Therefore, threats that have affected the Tortuguero population as described previously, may ultimately influence the other population trajectories, including Florida. Given the large size of the Tortuguero nesting population, which is currently in decline, its status and trend largely drives the status of North Atlantic DPS.

Aside from the long-term increasing nesting trend observed in Florida, the declining trend in nesting observed in Toruguero indicates a species in decline. However, since we anticipate 621 mortalities over the next 150 years, which is only a small 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.

Recovery

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

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

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

According to data collected from Florida's index nesting beach survey from 1989-2021, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 40,911 in 2019. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011. The pattern departed from the low lows and high peaks in 2020 and 2021 as well, when 2020 nesting only dropped by half from the 2019 high, while 2021 nesting only increased by a small amount over the 2020 nesting, with another increase in 2022 still well below the 2019 high. This overall increasing trend in nesting at Florida's index beaches indicates that the first listed recovery objective is being met. There are no estimates specifically addressing changes in abundance of individuals on foraging grounds currently available. 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 621 green sea turtles from the North Atlantic DPS over the next 150 years resulting from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area will cause 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 North Atlantic DPS green sea turtles' recovery in the wild even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion.

Conclusion
The lethal take of 621 green sea turtles from the North Atlantic DPS over the next 150 years resulting from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the North Atlantic DPS of green sea turtle in the wild.

8.2 Kemp's Ridley Sea Turtles

The deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area over a period of 10 years may result in the lethal take of 1,970 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 1,611 during the next 75 years (one turtle per year per structure deployed), and approximately 359 during the last 50 years (one turtle every 3 years per structure deployed).

Survival
The potential lethal take of up to 1,970 Kemp's ridley sea turtles over the next 150 years from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area as 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 take 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 1,970 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 the high-relief artificial reef material will occur opportunistically as materials and funding become available and deployments will occur only within a discrete area. 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). Nesting numbers rebounded in 2020 (18,068 nests) and 2021 (17,671 nests) (CONAMP data, 2021).

A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 [NPS data]. 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. Numbers rebounded again in 2020 with 262 nests, dropped in 2021 to 195 nests, then rebounded to 284 nests in 2022 (National Park Service data).

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 ridley sea turtles. 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. Recent years have seen nesting data plateau, and it is unknown whether the population is stabilizing or is likely to increase again.

While it is clear that the population has increased over the long-term, the future trajectory of nesting trends is unclear. We anticipate 1,970 mortalities of Kemp's ridley sea turtles over the next 150 years, which is only a small fraction of the oscillating 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.

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 2022 indicate there were a total of 17,418 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. 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 1,970 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.

Conclusion
The lethal take of 1,970 Kemp's ridley sea turtles over the next 150 years resulting from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the Kemp's ridley sea turtle in the wild.

8.3 Leatherback Sea Turtles

The deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area over a period of 10 years may result in the lethal take of 14 leatherback sea turtles over the next 150 years. The take is expected to be no leatherback sea turtles during the first 25 years, approximately 11 during the next 75 years (one turtle per year per structure deployed), and approximately 3 during the last 50 years (one turtle every 3 years per structure deployed).

Survival

The potential lethal take of up to 14 leatherback sea turtles over the next 150 years from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area 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 opportunistically as materials and funding become available and deployments will occur only within 3 discrete 1 nm2 areas (i.e., Fish Havens 20, 21, and 22). 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 30 leatherback sea turtle over the next 150 years accounts for only 0.003158-0.088235% 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 [excluding 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 Tortuguero, Costa Rica, 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 14 mortalities over the next 150 years, which is only a small 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.

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 Section 4.1.4, data from 2018 have 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 14 leatherback sea turtle over the next 150 years (with no takes anticipated during the first 25 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.

Conclusion
The lethal take of 14 leatherback sea turtles over the next 150 years resulting from the deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area 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.

8.4 Loggerhead Sea Turtles

The deployment of up to 50 high-relief artificial reef structures within the Sandy Reef deployment area over a period of 10 years may result in the lethal take of 1,970 loggerhead sea turtles in the Northwest Atlantic DPS over the next 150 years. The take is expected to be no loggerhead sea turtles during the first 25 years, approximately 1,611 during the next 75 years (one turtle per year per structure deployed), and approximately 359 during the last 50 years (one turtle every 3 years per structure deployed).

Survival
The potential lethal take of up to 1,970 loggerhead sea turtles in the Northwest Atlantic DPS over the next 150 years from the deployment of up 50 high-relief artificial reef structures within the Sandy Reef deployment area 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), 1,611 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 deployment of the high-relief artificial reef material will occur opportunistically as materials and funding become available and deployments will occur only within a discrete area. Loggerhead sea turtles in the Northwest Atlantic DPS generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether 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 4.1.5, we reviewed the status of this species in terms of nesting and female population trends and several 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 Northwest Atlantic DPS in the 2004-2008 timeframe to likely be between approximately 20,000-40,000 individuals (median 30,050), with a low likelihood of being as many as 70,000 individuals. Another estimate for the entire western North Atlantic population was a mean of 38,334 adult females using data from 2001-2010 (Richards et al. 2011). A much less robust estimate for total benthic females in the western North Atlantic was also obtained, with a likely range of approximately 30,000-300,000 individuals, up to less than 1 million.

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. Since the beginning of the index program in 1989, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that was followed by a sharp decline over the subsequent 9 years. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represented a new record for loggerheads on the core index beaches. While nest numbers subsequently declined from the 2016 high FWRI noted that the 2007-2021 period represents a period of increase. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but non-significant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening confidence intervals. Nesting at the core index beaches declined in 2017 to 48,033, and rose again each year through 2020, reaching 53,443 nests, dipping back to 49,100 in 2021, and then in 2022 reaching the second-highest number since the survey began, with 62,396 nests. It is important to note that with the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.) it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani, et al. 2019).

The proposed action could lethally take 1,970 loggerhead sea turtles in the Northwest Atlantic DPS 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 in 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. Further, the lethal take calculated represents an overestimate of potential take over 150 years. Actual take will depend on the number of high-relief artificial reef materials actually deployed, and lethal take will likely be minimized by the implementation of the Construction Conditions and Best Practices outlined in Sections 2.1.2 and 2.1.3.

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 Northwest Atlantic DPS of the loggerhead sea turtle in the wild.

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 1,970 loggerhead sea turtles in the Northwest Atlantic DPS over the next 150 years is minimal in relation to the overall population, and 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.

Conclusion

The lethal take of 1,970 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 Northwest Atlantic DPS of the loggerhead sea turtle in the wild.

9 CONCLUSION

We reviewed the Status of the Species, the Environmental Baseline, the Effects of the Action, and the Cumulative Effects using the best available data. The proposed action will result in the take of the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS). Given the nature of the proposed action and the information provided above, we conclude that the action, as proposed, is not likely to jeopardize the continued existence of the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS).

10 INCIDENTAL TAKE STATEMENT

10.1 Overview

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 (ESA Section 2(19)). *Incidental take* refers to takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant. 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 and the Terms and Conditions of the Incidental Take Statement of the Opinion.

Section 7(b)(4)(c) of the ESA specifies that to provide an Incidental Take Statement for an endangered or threatened species of marine mammal, the taking must be authorized under Section 101(a)(5) of the MMPA. Since no incidental take of listed marine mammals is anticipated as a result of the proposed action, no statement on incidental take of protected marine mammals is provided and no take is authorized. Nevertheless, the applicant must immediately notify (within 24 hours, if communication is possible) our Office of Protected Resources if a take of a listed marine mammal occurs.

As soon as the applicant becomes aware of any take of an ESA-listed species under NMFS's purview that occurs during the proposed action, the applicant shall report the take to NMFS SERO PRD via the NMFS SERO Endangered Species Take Report Form (https://forms.gle/85fP2da4Ds9jEL829). This form shall be completed for each individual known reported capture, entanglement, stranding, or other take incident. Information provided via this form shall include the title, Mexico Beach Sandy Reef Artificial Reef, the issuance date, and ECO tracking number, SERO-2023-00489, for this Opinion; the species name; the date and time of the incident; the general location and activity resulting in capture; 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. At that time, consultation may need to be reinitiated.

The USACE has a continuing duty to ensure compliance with the reasonable and prudent measures and terms and conditions included in this Incidental Take Statement. If the USACE (1) fails to assume and implement the terms and conditions or (2) fails to require the terms and conditions of the Incidental Take Statement through enforceable terms that are added to the permit or grant document or other similar document, the protective coverage of Section 7(o)(2) may lapse. 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 Incidental Take Statement (50 CFR 402.14(i)(3)).

10.2 Amount of Extent of Anticipated Incidental Take

NMFS anticipates the total lethal take over the next 150 years as a result of the project will consist of up to 621 green sea turtles (North Atlantic DPS), up to 1,970 Kemp's ridley sea turtles, 14 leatherback sea turtles, and 1,970 loggerhead sea turtles (Table 8).

Table 8. Anticipated Future Take by Species and DPS over 150 years.

Species	Estimated lethal take during first 25 years	Estimated lethal take during first 50 years	Estimated lethal take during first 75 years	Estimated lethal take during first 100 years	Estimated lethal take over entire 150 years
Green sea turtle (North Atlantic DPS)	0	169	339	508	621
Kemp's ridley sea turtle	0	537	1,074	1,611	1,970
Leatherback sea turtle	0	3	7	11	14
Loggerhead sea turtle (Northwest Atlantic DPS)	0	537	1,074	1,611	1,970

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 8. 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 8 for any defined time period will require reinitiation (i.e., take higher than 0 for any species during the first 25 years of life for any high-relief artificial reef structure placed will require reinitiation).

10.3 Effect of Take

NMFS has determined that the anticipated take specified in Section 10.2 is not likely to jeopardize the continued existence of green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, leatherback sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS) if the project is developed as proposed.

10.4 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires NMFS to issue to any federal agency whose proposed action is found to comply with Section 7(a)(2) of the ESA, but may incidentally take individuals of listed species, a statement specifying the impact of that taking. The Incidental Take Statement must specify the Reasonable and Prudent Measures necessary to minimize the impacts of the incidental taking from the proposed action on the species, and Terms and Conditions to implement those measures. "Reasonable and prudent measures" are measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take" (50 CFR 402.02). Per Section 7(o)(2), any incidental taking that complies with the specified terms and conditions is not considered to be a prohibited taking of the species concerned.

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

NMFS has determined that the following Reasonable and Prudent Measures are necessary and appropriate to minimize impacts of the incidental take of ESA-listed species related to the proposed action. The following Reasonable and Prudent Measures and associated terms and conditions are established to implement these measures, and to document incidental takes. Only incidental takes that occur while these measures are in full implementation are not considered to be a prohibited taking of the species. These restrictions remain valid until reinitiation and conclusion of any subsequent Section 7 consultation.

1. The USACE must ensure that the applicant provides take reports regarding all interactions with ESA-listed species at Sandy Reef.
2. The USACE must ensure that the applicant minimizes the likelihood of injury or mortality to ESA-listed species resulting from entanglement in lost fishing gear or marine debris that accumulates at Sandy Reef.
3. The USACE must ensure that the applicant coordinates periodic marine debris removal (i.e., cleanup) events concurrent with required annual monitoring at Sandy Reef.

10.5 Terms and Conditions

In order to be exempt from the prohibitions established by Section 9 of the ESA, the USACE must comply (or must ensure that any applicant complies) with the following Terms and Conditions.

The following Terms and Conditions implement Reasonable and Prudent Measure #1:

- If the applicant 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, Jacksonville District Engineer so that any necessary stranding response coordination can be initiated with the U.S. Fish and Wildlife Service or National Marine Fisheries Service.
- The USACE must ensure that the applicant reports all known captures of ESA-listed species and any other takes of ESA-listed species to the NMFS SERO PRD.
- If and when the applicant becomes aware of any known reported capture, entanglement, stranding, or other take, the applicant must report it to NMFS SERO PRD via the NMFS SERO Endangered Species Take Report Form (https://forms.gle/85fP2da4Ds9jEL829).
    - This form must reference this Opinion by the NMFS tracking number (SERO-2023-00489 Mx Bch Sandy Reef AR) and date of issuance.
    - This form shall be completed for each individual known reported capture, entanglement, stranding, or other take incident.
    - Information provided via this form shall include the species name; the date and time of the incident; the general location and activity resulting in capture; 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.
- Every year, the applicants must submit a summary report of capture, entanglement, stranding, or other take of ESA-listed species at Sandy Reef to NMFS SERO PRD by email: nmfs.ser.esa.consultations@noaa.gov.

- Emails and reports must reference this Opinion by the NMFS tracking number (SERO-2023-00489 Mx Bch Sandy Reef AR) and the date of issuance.
- The report will contain the following information: the total number of ESA-listed species captures, entanglements, strandings, or other take that was reported at Sandy Reef.
- The report will contain all information for any sea turtles taken to a rehabilitation facility holding an appropriate USFWS Native Endangered and Threatened Species Recovery permit. This information can be obtained from the appropriate State Coordinator for the STSSN (https://www.fisheries.noaa.gov/state-coordinators-sea-turtle-stranding-and-salvage-network)
- The first report will be submitted by January 31 of the year following issuance of the permit and will cover the period from permit issuance through December 31 of that year. The second report will be submitted by January 31 of the following year, and will cover the previous calendar year and the information in the first report. Thereafter, reports will be prepared every year, covering the prior rolling three-year time period, and emailed no later than January 31 of any year.
- Reports will include records of the clean-ups required in the terms and conditions in 3, below.

The following Terms and Conditions implement Reasonable and Prudent Measure #2:

- The USACE must ensure that the applicant provides to the public educational resources on reducing marine debris along with all physical and online promotional materials for the Bay County artificial reefs. Examples are available at https://marinedebris.noaa.gov/multimedia/posters

The following Terms and Conditions implement Reasonable and Prudent Measures #2 and #3:

- The USACE must ensure that the applicant will:
  - Conduct in-water structure cleanups on a regular basis to remove any derelict tackle, fishing line, or marine debris attached to the structure.
  - Submit a record of each cleaning event in the report required by T&C 1 above.

11 CONSERVATION RECOMMENDATIONS

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

- Conduct or fund research designed to increase the public's knowledge and awareness of marine debris and its impacts on ESA-listed species.

- Provide funding or resources (e.g., divers, equipment, etc.) to aid annual monitoring and frequent reef clean-ups to prevent the accumulation of lost fishing gear and marine debris.

12 REINITIATION OF CONSULTATION

This concludes formal consultation on the proposed action. As provided in 50 CFR 402.16, reinitiation of formal consultation is required and shall be requested by USACE or by the Service, where discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if: (a) the amount or extent of incidental take specified in the Incidental Take Statement is exceeded, (b) new information reveals effects of the action on listed species or critical habitat in a manner or to an extent not considered in this Opinion, (c) the action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this Opinion, or (d) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the USACE must immediately request reinitiation of formal consultation and project activities may only resume if USACE establishes that such continuation will not violate Sections 7(a)(2) and 7(d) of the ESA.

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