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

10/18/2019

F/SER31:SBC
SERO-2019-00225, SER-2019-19783

Chief, Washington Regulatory Field Office
Wilmington District Corps of Engineers
Department of the Army
2407 W 5th Street
Washington, North Carolina 27889

Dear Mr. Pelletier:

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

The Opinion considers the effects of the U.S. Army Corps of Engineers (USACE) Wilmington District permitting the placement of suitable reef building materials within the boundaries of 43 existing ocean reefs and 25 existing estuarine reefs off North Carolina that are maintained by the state of North Carolina by the North Carolina Division of Marine Fisheries (NCDMF), as well as the construction of up to 5 new ocean reefs and 10 new estuarine reefs over the next 7 years. This time frame corresponds with the duration of the existing and planned USACE Wilmington District regional and programmatic general permits issued for the construction, maintenance and repair of artificial reefs. We base this Opinion on project-specific information provided in the consultation package as well as NMFS's review of published literature. This Opinion analyzes the potential for the project to affect the following species: green sea turtle (North Atlantic and South Atlantic distinct population segments [DPSs]), hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), shortnose sturgeon, Atlantic sturgeon (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs), giant manta ray, oceanic whitetip shark, blue whales, sei whales, sperm whales, fin whales, and North Atlantic right whales. This Opinion also analyzes the potential for the project to have an effect on the following designated critical habitat: loggerhead sea turtle (Northwest Atlantic DPS) (Constricted Migratory Habitat [Unit N-01], Migratory/Winter Habitat [Unit N-02], and Nearshore Reproductive Habitat [Unit N-03]), Atlantic sturgeon Critical Habitat, and North Atlantic right whale Critical Habitat (Unit 2). Pursuant to 50 CFR 402.14(i)(5), any taking which is subject to an incidental take statement (ITS), and which is in compliance with the terms and conditions (T&C) associated with the ITS, is not a prohibited taking under the ESA.

Please direct any questions to Shelby Creager, Consultation Biologist, by phone at (727) 209-5951, or by email at Shelby.Creager@noaa.gov.

Sincerely,

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for Roy E. Crabtree, Ph.D.
Regional Administrator

Enclosures:
Biological Opinion

File: 1514-22 f.1

Endangered Species Act - Section 7 Consultation
Biological Opinion

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

Applicant: North Carolina Division of Marine Fisheries

Permit Number SAW-2018-01538

Activity: North Carolina Division of Marine Fisheries Artificial Reef Program

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

Consultation Number SER-2019-19783, SERO-2019-00225

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Approved by:

for Roy E. Crabtree, Ph.D., Regional Administrator
NMFS, Southeast Regional Office
St. Petersburg, Florida

Date Issued: 10/18/2019

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Acronyms and Abbreviations

CFR	Code of Federal Regulations
CCL	Curved Carapace Length
DPS	Distinct Population Segment
DTRU	Dry Tortugas Recovery Unit of loggerhead sea turtle
DWH	Deepwater Horizon oil spill of 2010
EEZ	Exclusive Economic Zone
ESA	Endangered Species Act
FR	Federal Register
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
GADNR	Georgia Department of Natural Resources
GCRU	Greater Caribbean Recovery Unit of loggerhead sea turtle
ITS	Incidental Take Statement
MSA	Magnuson-Stevens Fishery Conservation and Management Act
NA DPS	North Atlantic DPS of green sea turtle
NCDMF	North Carolina Division of Marine Fisheries
NCWRC	North Carolina Wildlife Resources Commission
NGMRU	Northern Gulf of Mexico Recovery Unit of loggerhead sea turtle
NMFS	National Marine Fisheries Service
NOAA	National Ocean and Atmospheric Association
NRU	Northern Recovery Unit of loggerhead sea turtle
NWA DPS	Northwest Atlantic DPS of loggerhead sea turtle
Opinion	Biological Opinion
PCE	Primary Constituent Elements
PCTS	Public Consultation Tracking System
PRD	NMFS Southeast Regional Office, Protected Resources Division
RPM	Reasonable and Prudent Measure
SA DPS	South Atlantic DPS of green sea turtle
SARBO	South Atlantic Regional Biological Opinion
SAV	Submerged Aquatic Vegetation
SCDNR	South Carolina Department of Natural Resources
SGFMP	Snapper-Grouper Fishery Management Plan
STSSN	Sea Turtle Stranding and Salvage Network
T&C	Terms and Conditions
U.S.	United States of America

USACE	U.S. Army Corps of Engineers
USCG	U.S. Coast Guard
USFWS	U.S. Fish and Wildlife Service
USN	U.S. Navy

Units of Measurement

°C	degree Celsius
cm	centimeters
CPUE	catch per unit effort
ft	foot/feet
g	gram(s)
in	inch(es)
kg	kilogram(s)
km	kilometer(s)
lb	pound(s)
nmi^2	nautical square mile(s)
m	meter(s)
mm	millimeter(s)
mi	mile(s)
SCL	straight carapace length

INTRODUCTION

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. §1531 et seq.), requires that each federal agency ensure that any action authorized, funded, or carried out by the 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 those species. When the action of a federal agency may affect a protected species or its critical habitat, that agency is required to consult with either National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the protected species that may be affected.

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

This document represents NMFS's Opinion based on our review of the effects of placing additional suitable reef building materials within the boundaries of 43 existing ocean reefs and 25 existing estuarine reefs maintained by the state of North Carolina by the North Carolina Division of Marine Fisheries (NCDMF), as well as the construction of up to 5 new ocean reefs and 10 new estuarine reefs over the next 7 years. This Opinion analyzes the potential for the project to affect the following species: loggerhead sea turtle (Northwest Atlantic distinct population segment [DPS]), green sea turtle (North Atlantic and South Atlantic DPSs), Kemp's ridley sea turtle, leatherback sea turtle, hawksbill sea turtle, shortnose sturgeon, Atlantic sturgeon (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs), giant manta ray, oceanic whitetip shark, blue whales, sei whales, sperm whales, fin whales, and North Atlantic right whale. This Opinion also analyzes the potential for the project to have an effect on the following designated critical habitat: loggerhead sea turtle (Northwest Atlantic DPS) critical habitat (Constricted Migratory Habitat [Unit N-01], Migratory/Winter Habitat [Unit N-02], and Nearshore Reproductive Habitat [Unit N-03]), North Atlantic right whale critical habitat (Unit 2), and Atlantic sturgeon critical habitat. NMFS has analyzed the information provided by the USACE and the effects on listed species under our purview in accordance with Section 7 of the ESA of 1973, as amended (16 U.S.C. 1531 et seq.). NMFS bases this Opinion on project information provided by the U.S. Army Corps of Engineers (USACE) as well as published literature and the best available scientific and commercial information.

Updates to the regulations governing interagency consultation (50 CFR part 402) will become effective on October 28, 2019 [84 FR 44976]. Because this consultation was pending and will be completed prior to that time, we are applying the previous regulations to the consultation. However, as the preamble to the final rule adopting the new regulations noted, "[t]his final rule does not lower or raise the bar on section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and consistency, streamlines consultations, and codifies existing practice." Thus, the updated regulations would not be expected to alter our analyses.

1. CONSULTATION HISTORY

The following is the consultation history for identifier number SERO-2019-00225, NCDMF Artificial Reef Programmatic. This project was originally assigned a Public Consultation Tracking System (PCTS) number (SER-2019-19783) in our now obsolete tracking system. The project was assigned a tracking number in our new NMFS Environmental Consultation Organizer (ECO), SERO-2019-00225.

- On June 13, 2018, USACE Wilmington emailed Consultation Biologist Shelby Creager a rough draft of the Programmatic effort as an informal expedited document for pre-technical assistance and feedback before submitting for a consultation request.
- On January 29, 2019, NMFS received a request for informal consultation under Section 7 of the ESA from the USACE for permit SAW-2018-01538 in a letter dated December 20, 2018. The consultation was held in abeyance for 38 days due to a lapse in appropriations and resulting partial government shutdown.
- NMFS and USACE corresponded via telephone on March 7, 2019, and via email on March 22, 2019, to discuss project details and request additional information. USACE responded via email on March 28 and April 11, 2019.
- NMFS and USACE corresponded via conference call on April 11, 2019, to discuss issues with the Project Design Criteria (PDCs) and questions from NMFS's Request for Additional Information.
- On May 6, 2019, NMFS requested additional information with questions relating to the Bonner Bridge demolition and high relief material, NCDMF responded on the same day, and consultation was initiated that day.
- On June 14, 2019, NMFS requested additional information with questions relating to the proposed action, PDCs, and permit details. NCDMF responded on the same day.
- On July 1, 2019, NMFS requested additional information on the accuracy of drafts of the proposed action, existing materials, and PDCs. Consultation was submitted for internal review.
- On July 11, 2019, USACE responded to NMFS request for additional information with additional questions requiring NMFS' response. NMFS responded on August 28, 2019.
- On September 12, 2019, USACE sent NMFS a separate consultation request (SERO-2019-02609, SAW-2017-01724) for an artificial reef enhancement project (AR-197), and NMFS confirmed with USACE via phone call on October 11, 2019, that site AR-197 is one of the 25 existing estuarine reefs evaluated under the proposed action of this Opinion.
- On September 13, 2019, consultation was submitted to General Counsel for review.
- On October 4, 2019, NMFS sent USACE a request for additional information clarifying programmatic language. NMFS received a response on October 16, 2019, and sent USACE a request to add clarifications to the proposed action and annual reporting requirements. On October 17, 2019, USACE confirmed the clarifications to the proposed action and annual reporting requirements.

2. DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA

This Programmatic Opinion evaluates the placement of additional suitable reef building materials within the boundaries of 43 existing ocean reefs and 25 existing estuarine reefs maintained by the North Carolina Division of Marine Fisheries (NCDMF), as well as the construction of up to 5 new ocean reefs and 10 new estuarine reefs over the next 7 years. The USACE authorizes this activity using their General Permit Numbers 198000291 and 198500194. These are general permits to perform work in or affecting navigable waters of the United States, pursuant to Section 10 of the Rivers and Harbors Act (RHA) of March 3, 1899, (33 U.S.C. 403), Section 404 of the Clean Water Act (CWA) (33 U.S.C. 1344), and Section 4(e) of the Outer Continental Shelf Lands Act (OCSLA) of 1953 [43 U.S.C. 1333(e)]. This Opinion only covers the artificial reef activities described in this Opinion and does not cover all types of activities or projects that the USACE permits under its CWA RHA, or OCSLA authority.

The goal of this Opinion is to streamline and consolidate ESA Section 7 consultation for the majority of artificial reef construction and deployment activities that the USACE Wilmington District authorizes off the coast of North Carolina. In this Opinion, we are able to provide a more comprehensive and cohesive review of the majority of the artificial reef construction projects in North Carolina that are expected to be permitted by the USACE and to analyze the cumulative effects of these actions, than if we consulted on a project-by-project basis.

Below, we provide a description of:

- How the Opinion may be used.
- The artificial reef construction activities covered by this Opinion, including the PDCs those activities must meet to be covered (Section 2.1).
- An estimate of the number of artificial reef deployments and construction activities that the USACE will permit over a 7-year period that are covered by this Opinion (Section 2.1).
- The areas in North Carolina in which the permitted activities covered under this Opinion occur, including areas where additional restrictions apply (Section 2.2).
- The project-specific review (Section 2.3) and programmatic review (Section 2.4) requirements necessary to ensure that the reliance on this Opinion is limited to those actions in North Carolina that meet the PDCs in this Opinion and are consistent with this Opinion.

Use of the Opinion

As explained above, the USACE requested consultation on the effect of authorizing the NCDMF to place additional suitable reef building materials within the boundaries of 43 existing ocean reefs and 25 existing estuarine reefs maintained by the state of North Carolina. The construction of new artificial reefs may be proposed by the NCDMF and up to five ocean reefs (in state and/or federal waters) and ten estuarine reefs may be constructed over the next 7 years. The USACE determined that the action may affect certain ESA-listed species or designated critical habitat within NMFS's purview (see Section 3 for the USACE's effects determinations). NMFS believes that all of the activities described below, if carried out as described, will have effects on

ESA-listed species and/or critical habitat as discussed herein. The USACE may rely on this Opinion to meet its ESA Section 7 consultation requirements when authorizing activities that meet the PDCs and other requirements of the Opinion. Nothing in this Opinion precludes the USACE from determining that a future project does not affect an ESA-listed species or designated critical habitat. NMFS and the USACE will continue to discuss the Opinion as it is applied, at the project-specific and programmatic reviews described in Sections 2.3 and 2.4, and may refine it in the future.

Two USACE Divisions, the USACE Regulatory Division and the USACE Civil Works Division, have responsibility for authorizing and/or implementing in-water projects under the CWA and the RHA and consulting on the effects of those projects under the ESA. The USACE Regulatory Division can satisfy their ESA Section 7 requirements by relying on this Opinion for projects meeting the requirements of this Opinion.

2.1. Proposed Action

The USACE proposes to authorize the NCDMF to place additional suitable reef building materials within the boundaries of 43 existing ocean reefs and 25 existing estuarine reefs maintained by the state of North Carolina. The construction of new artificial reefs may be proposed by the NCDMF and up to five ocean reefs (in state and/or federal waters) and ten estuarine reefs may be constructed over the next 7 years (current permit expires in 2021 and NCDMF will renew for another 5 years) dependent upon available funding, suitable materials, appropriate sites, etc.

The USACE proposes to deploy low-relief and high-relief materials over a 7-year period for the creation of 5 new ocean reefs and 10 new estuarine reefs. Low-relief reef material includes materials such as solid concrete material, rock rubble, and individual artificial reef modules that present less complicated vertical relief. NMFS considers high-relief, complex artificial reef material to include any vessel, aircraft, decommissioned oil rig, bridge span, metal tower, etc., that extends 7 feet (ft) or more from the seafloor and that has a footprint greater than 200 square feet (ft^2) (individually or collectively), excluding prefabricated artificial reef modules.

Reef materials will include (but are not limited to) oval and round pipe, box culverts, risers, catch-basins, knock-out boxes, bridge rubble and bridge spans, manholes, slabs, pilings, crushed concrete, natural materials, and pre-designed structures such as Reef Balls or other units. Prefabricated reef modules will consist of any hard, strong building material made by mixing a cementing material (commonly Portland cement) and a mineral aggregate with sufficient water to cause material to set and bind. Vessels may also be utilized as artificial reef building material under the proposed action. The types and locations of reef material placement depends on funding, material availability, and involvement of stakeholders. Based on the best available estimate at this time, the anticipated reef sites for high relief vessels and low relief concrete and aggregate rock that may be deployed over the next 7 years are listed in Table 1.

Table 1. Future NCDMF Reef Projects Anticipated through 2026*

Year	Anticipated Reef Sites	Material
2019	AR-165, AR-250, AR-255, AR-368	4 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2020	AR-368, AR-430	2 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2021	AR-430, AR-305	4 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2022	AR-305	3 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2023	Unknown	3 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2024	Unknown	3 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2025	Unknown	3 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock
2026	Unknown	3 vessels, 15,000 tons recycled concrete, 37,500 tons aggregate rock

*additional low relief material may be added to existing and new sites during this time frame

2.1.1 Artificial Reef Sites

Existing artificial reef sites include estuarine reefs (Figure 1), coastal ocean reefs (Figure 2), and ocean reefs (Figure 3) that range from 30 ft to 160 ft deep and are located inshore to as far out as 28.89 nautical miles (nmi). The offshore reef sites are circular and are delineated as a coordinate and a radius distance. Of the 43 coastal and ocean reef sites, 41 have a 1,500-ft radius (162.27 acres [ac]), the remaining two have a 3,000-ft radius (649.09 ac). The material distribution in each site is variable. Some sites have densely packed material, while others have small patches distributed throughout. The new artificial reef sites proposed are anticipated to be similar in the size, depth, and distance from shore as existing reefs.

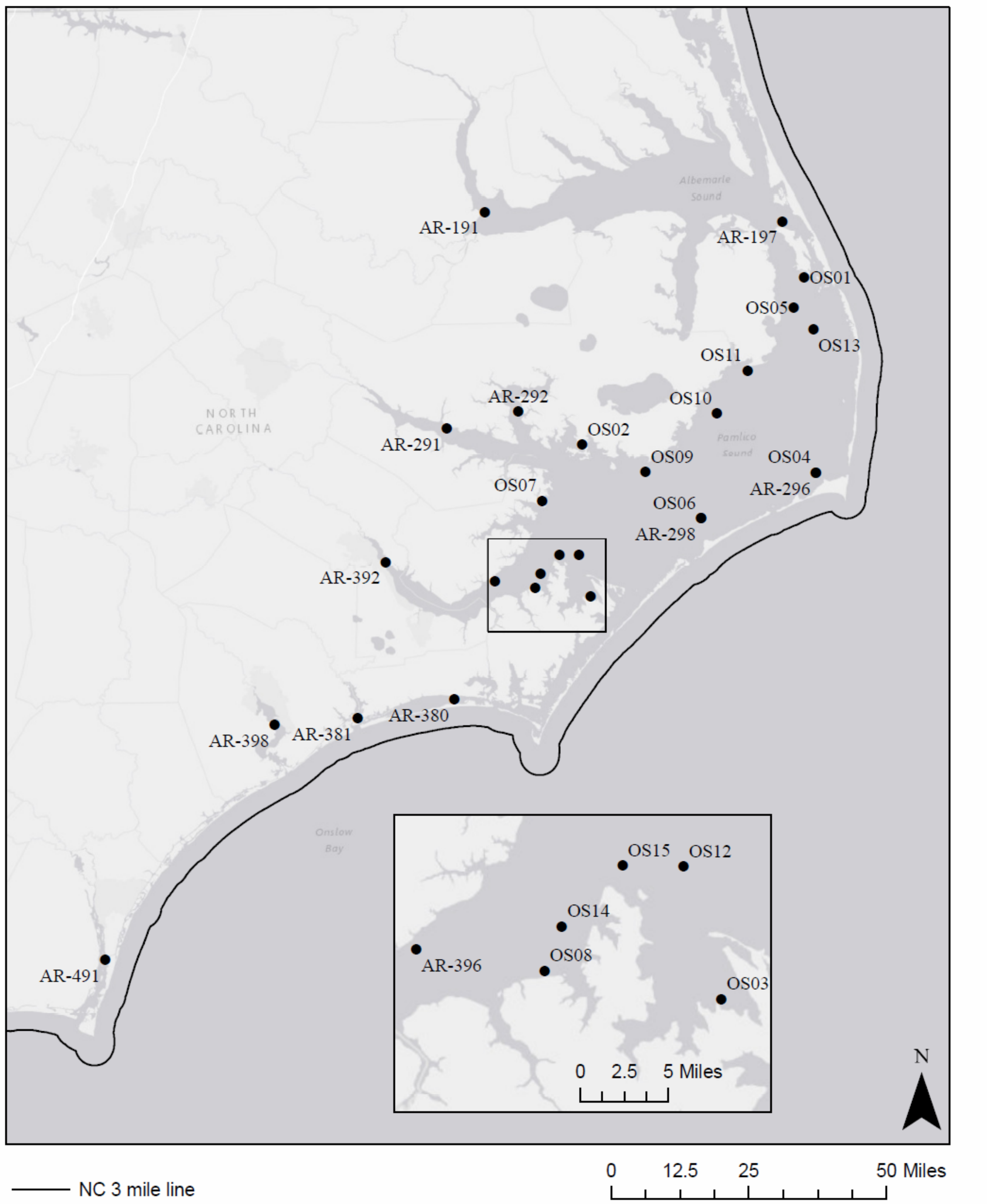

Figure 1. Existing North Carolina Estuarine Reefs (provided by USACE)

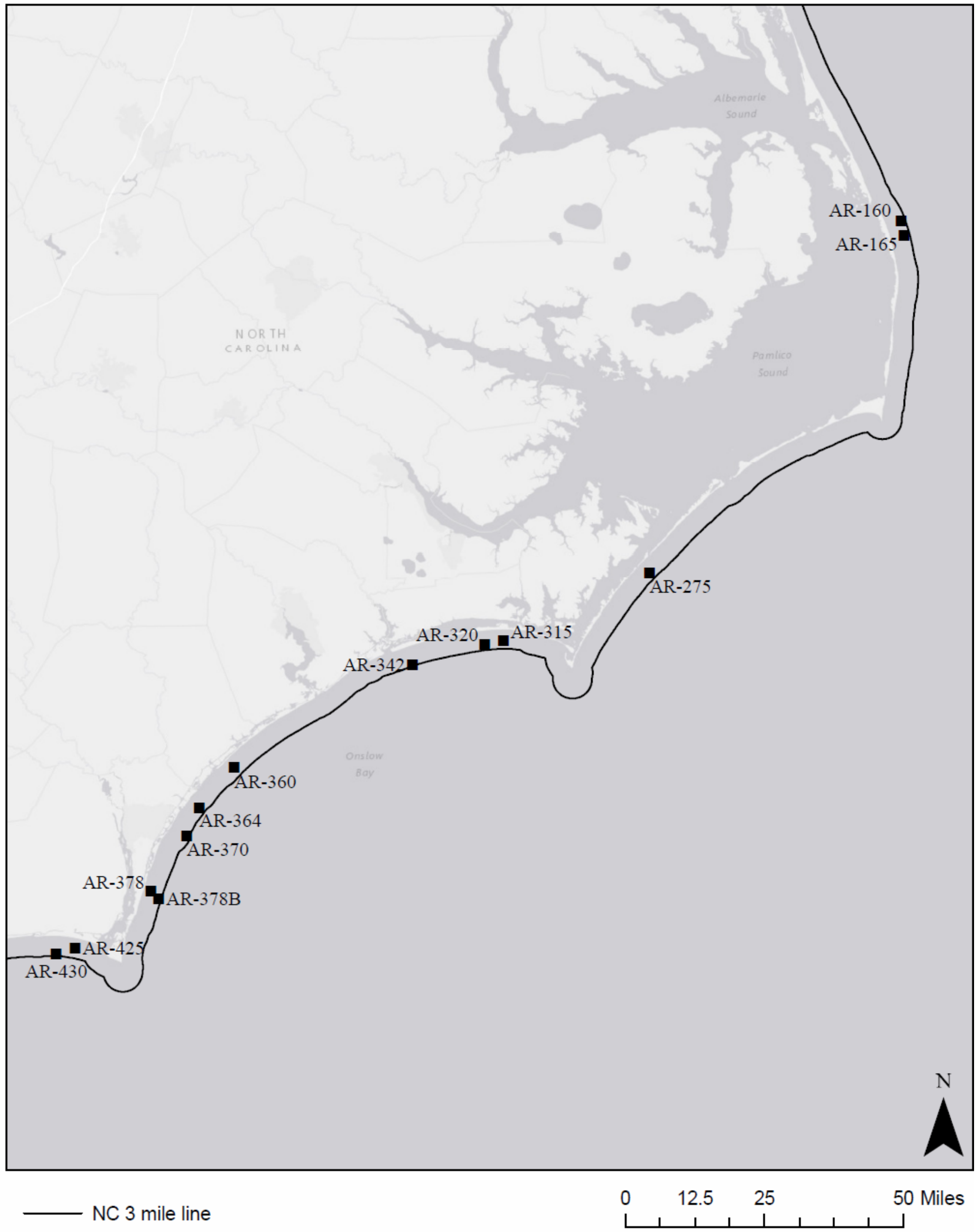

Figure 2. Existing North Carolina Coastal Ocean Reefs (provided by USACE)

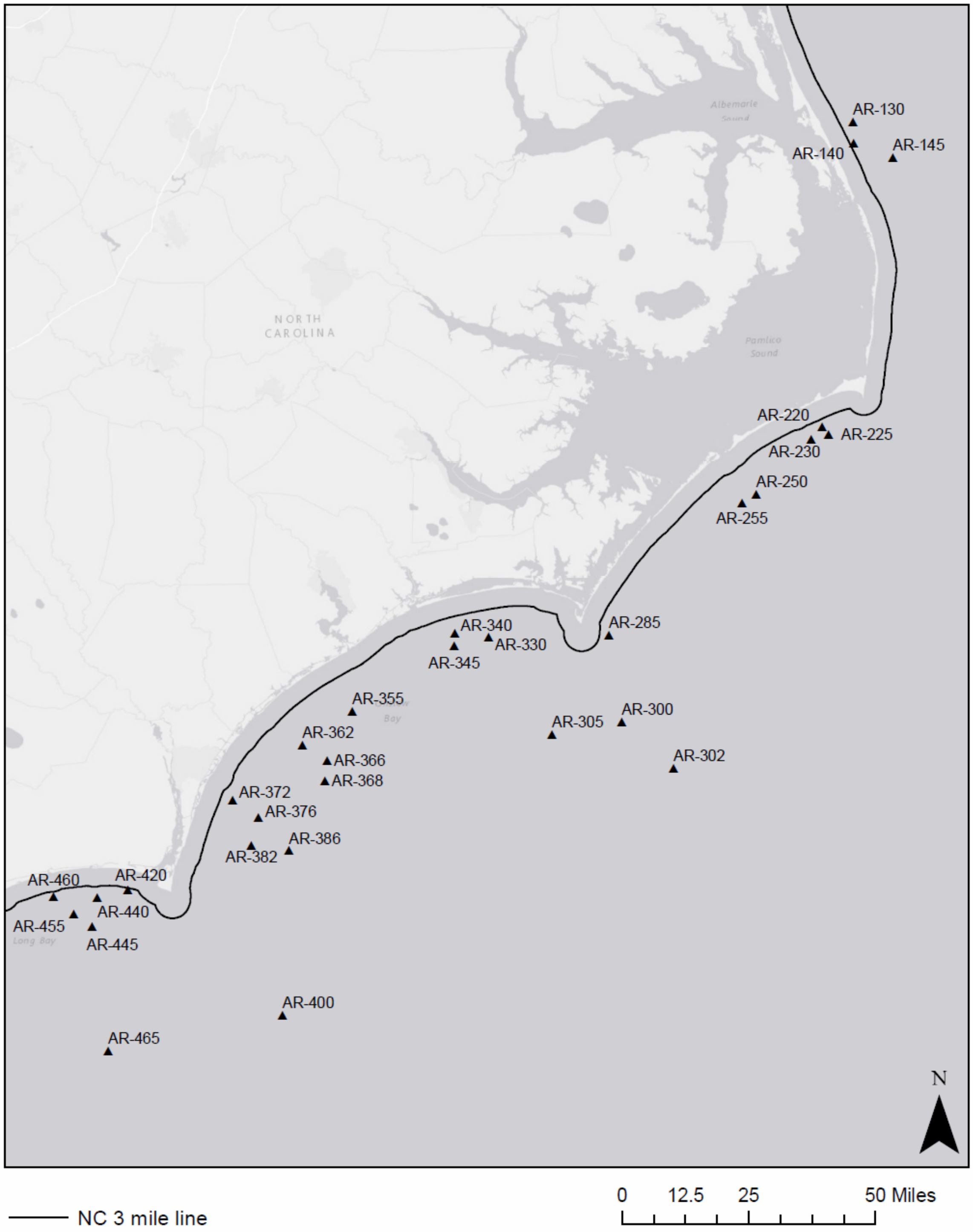

Figure 3. Existing North Carolina Federal Ocean Reefs (provided by USACE)

2.1.2 Artificial Reef Construction and Deployment

Reef building material will be stockpiled and retained within an upland location, transferred to a barge by heavy equipment, and transported to the reef location. Materials will be dropped in the water by excavators, loaders, or other heavy equipment via barge and allowed to sink. Reef construction times will vary upon amount of material, weather, and other site/species-specific guidelines/regulations. Materials will be placed by NCDMF staff or its contractors.

During estuarine deployment, temporary buoys will be used to mark locations for placement (permanently marked per United States Coast Guard (USCG) Aid to Navigation requirements). A 150-foot buffer of no construction will be maintained inside the perimeter of the reef boundary and the remaining reef will be gridded into 150-ft x 150-ft grids for planning and spatial reference. Estuarine reefs are typically marked on all four corners with buoys or pilings. Buoys are connected to anchors with 3/8-inch (in) galvanized chain and shackles. Reefs marked with pilings will use no more than 3-pile dolphins on each corner, for a total of 12 pilings. Reefs in lower energy environments may be marked with fewer or smaller pilings. All piles will be pressure treated wood, large piles will be installed using a vibratory hammer and small piles may be washed down with a pump. Reefs in the ocean will not be marked with buoys or pilings.

The NCDMF will consider the development of the 5 new ocean reefs and 10 new estuarine reefs analyzed in this Opinion when funding and resources become available. When siting new reefs the NCDMF will avoid natural hard bottom, submerged aquatic vegetation, shell resources, or other unknown objects, such as existing reef material. This is verified by a variety of methods including side-scan survey, SCUBA surveys, visual inspections, and/or benthic grabs. Permittees must adhere to conditions set forth in the Corps' Regional General Permit 198500194 and Programmatic General Permit 1980002911. Construction schedules and methods will also follow PDCs for protected species described in Appendix A.

Underwater reef cleanups will be performed by NCDMF divers at some sites during annual reef site monitoring events. During regularly scheduled reef assessments, staff will make reasonable attempts to clean reef materials of debris without compromising the safety of the crew and divers, as some sites prohibit cleanups due to dive safety restrictions. For example, precluding conditions might involve low visibility or high current, which may present diver safety issues and increase the potential of divers being entangled or snagged. The Permittee will send confirmation of cleanup to NMFS's Southeast Regional Office (takereport.nmfsser@noaa.gov), including dates of cleanup efforts and results of the cleanup.

2.2. Action Area

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02). For the purposes of this consultation, the Corps has defined the action area to include the inshore and offshore waters of the Atlantic Ocean as well as estuarine waters along the coast of North Carolina as shown in Figures 1, 2, and 3 above. Currently, the open waters of the Atlantic Ocean support recreational and commercial activity. There are a number of existing inshore and offshore artificial reefs located in nearby waters which are managed by the NCDMF. North Carolina's offshore reef sites are circular and are delineated as a coordinate and a radius distance.

Of the 43 reef sites, 41 have a 1500 ft radius (162.27 ac); the remaining two have a 3000 ft radius (649.09 ac). The material distribution in each site is variable. Some sites have densely packed material, while others have small patches distributed throughout. Estuarine sites are all built in sand, mud, and muddy sand without submerged aquatic vegetation (SAV) or shellfish resources. Ocean sites are built on hard sand bottom devoid of any natural limestone. There are currently 7 artificial reef sites located in North Atlantic right whale critical habitat, and 4 reef sites located in Atlantic sturgeon critical habitat, and many reefs located in North Atlantic Distinct Population Segment (DPS) of loggerhead sea turtle critical habitat. Additional information on the reef locations can be found at: http://portal.ncdenr.org/web/mf/artificial-reefs-program.

2.3. Project Specific Review

Before USACE can authorize a covered activity and rely on this Opinion to fulfill its consultation obligations under Section 7 of the ESA, the USACE must conduct a project-specific review to ensure that all of the PDCs are met. If the PDCs are met, then the project qualifies for coverage under the Opinion. Thereafter, the USACE must attest to, and certify, compliance with the PDCs and the Opinion in a submission to NMFS via the following email address: nmfs.ser.esa.consultations@noaa.gov. All pertinent PDCs will be included as a Special Condition of the Permit. The USACE remains responsible for conducting the programmatic review, discussed in Section 2.4.

<u>Superseding Process for Review and Inclusion of Substantially Similar Projects or Projects with Substantially Similar Effects</u>: In a few instances, NCDMF propose to use materials or installation methods that were not specifically considered in this Opinion, or the project may deviate from the PDCs in a minor fashion. For example, NCDMF may propose to use a different reef material than that considered in the Opinion. In those instances, the USACE must determine whether the effects of the modification on ESA-listed species or designated critical habitat are substantially similar to the effects considered in this Opinion. If the USACE makes that preliminary determination, it must provide that rationale to NMFS and request permission to rely on the Opinion to satisfy its ESA Section 7 consultation obligations. If NMFS determines that the effect is substantially similar to the effects discussed and found in this Opinion, then NMFS may approve the modification, on that case-specific basis alone, and the project can be covered under this Opinion, and will be included in the programmatic review. If the USACE seeks to authorize a project that proposes to use the same modification in the future, it still must seek permission for the modification. When requesting consideration for a new material, method, or modification, the USACE must await written approval from NMFS before authorizing the project. This process supersedes the review process described above.

2.4. Programmatic Review

NMFS and the USACE will conduct an annual programmatic review of the projects authorized, that rely on this Opinion to evaluate (1) whether the predicted nature and scale of the effects continues to be accurate; (2) whether the PDCs continue to avoid and minimize effects to species and critical habitat as designed or require modification; and (3) whether the project-specific review procedures are being followed and are effective at screening out projects that do not meet the PDCs or are not in compliance with the Opinion.

NMFS and the USACE's Wilmington District have dedicated project managers responsible for implementation, management, and administration of this Programmatic Opinion. The programmatic team leads from NMFS and the USACE, and other members as necessary, will participate in the annual programmatic review to verify conclusions regarding the potential effects to ESA-listed species and critical habitat, review data on the cumulative effects of the combined projects from the previous year(s), and evaluate and suggest any procedural changes (e.g., modifications to the reporting form or clarification on a PDC) prompted by the review of data. If the annual programmatic review shows that the anticipated effects to listed species or critical habitat, as discussed in this Opinion, are different than the effects of the projects as implemented, reinitiation of consultation may be required (50 CFR 402.16). Reviews will be conducted in the following way:

<u>Annual Review</u>: Each year, NMFS and the USACE will conduct an annual review of the projects authorized in reliance on this Opinion. The first annual review will cover projects authorized in the 12-month period starting from the date NMFS completes this Opinion. The second annual review will cover all projects authorized in the second year, and so on. The annual review consists of a USACE-data gathering and review component resulting in an annual spreadsheet from the USACE to NMFS and a NMFS review and comment on the USACE spreadsheet. A more specific description of the annual review process is described below.

The USACE shall provide NMFS with a completed Excel spreadsheet of all activities authorized using this Opinion, which will include the following: project dates; location; depth; bottom type; size of reef site, sections, and any gaps if required; material type; amount of material to be deployed; and transport methods. Before submitting the spreadsheet to NMFS, USACE shall check the spreadsheet for accuracy (e.g., properly formatted, completely filled out, no duplicates, latitude/longitude data is accurate and entered according to the formatting requirements provided). In addition, the USACE should review the data to determine whether the projects authorized are consistent with the Opinion (e.g., the PDCs are being met) and to confirm that its assumptions about the number and location of the projects (and any other assumptions that formed the basis of the effects analysis) were accurate. For example, this Opinion assumes that a specific number of activities will be completed over a 7-year period. If the data shows that the number of projects is likely to exceed that expected number, the USACE should inform NMFS in the annual review. Any lessons learned or procedural changes the USACE believes are necessary to improve the program shall also be part of this review. The USACE shall provide a short summary of its findings with its submission of the spreadsheet.

NMFS will review the data the USACE submits and provide comments or set up a conference call if needed to discuss the results. The NMFS and USACE programmatic team will then work to resolve any outstanding questions or concerns and the results of these discussions will be documented.

Following the annual review, NMFS and the USACE may jointly determine that revisions to the Opinion or the PDCs are necessary. Reinitiation of consultation may be required as appropriate as provided in 50 CFR Section 402.16.

3. STATUS OF LISTED SPECIES AND CRITICAL HABITAT

Table 2 and Table 3 provide the effect determinations for ESA-listed species and critical habitat the USACE and/or NMFS believe may be affected by the proposed action.

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

Species	ESA Listing Status[1]	Action Agency Effect Determination	NMFS Effect Determination
Sea Turtles			
Green (North Atlantic [NA] distinct population segment [DPS])	T	NLAA	LAA
Green (South Atlantic [SA] DPS)	T	NLAA	LAA
Kemp's ridley	E	NLAA	LAA
Leatherback	E	NLAA	LAA
Loggerhead (Northwest Atlantic [NWA] DPS)	T	NLAA	LAA
Hawksbill	E	NLAA	LAA
Fish			
Shortnose sturgeon	E	NLAA	NLAA
Atlantic sturgeon (All DPSs)	T/E[2]	NLAA	NLAA
Giant manta ray	T	NLAA	NLAA
Oceanic whitetip shark	T	NLAA	NLAA
Marine Mammals			
North Atlantic right whale	E	NLAA	NLAA
Blue whale	E	--	NLAA
Fin whale	E	--	NLAA
Sei whale	E	--	NLAA
Sperm whale	E	--	NLAA

Because adult Atlantic sturgeon from all DPSs mix extensively in marine waters, we expect fish from all 5 DPSs to be present in the action area.

[1] E = endangered; T = threatened; NLAA = may affect, not likely to adversely affect; NE = no effect; NP = not present
[2] The New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs are listed as endangered; the Gulf of Maine DPS is listed as threatened.

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

Species	Critical Habitat Unit	USACE Effect Determination	NMFS Effect Determination
Loggerhead sea turtle (NWA DPS)	Constricted Migratory Habitat (Unit LOGG-N-01), Winter Habitat (Unit LOGG-N-02), and Nearshore Reproductive (Unit LOGG-N-03)	NLAA	NLAA
Atlantic sturgeon (Carolina DPS)	Carolina Unit 2 Carolina Unit 3 Carolina Unit 4	NLAA	NLAA
North Atlantic right whale	Unit 2	NLAA	NLAA
NLAA = may affect, not likely to adversely affect			

3.1. Potential Routes of Effect Not Likely to Adversely Affect Listed Species

We have determined that the following potential routes of effect of the proposed action being considered in this Opinion are not likely to adversely affect any listed species or critical habitat in Tables 2 and 3. The following discussion summarizes our rationale for these determinations.

3.1.1 Effects of Material Deployment

ESA-listed sea turtles, fish, and marine mammals could be physically injured if struck by transport vessels or material during deployment at reef sites. We believe this effect will be discountable for the following reasons. All of these animals are highly mobile, and able to avoid slow-moving equipment. The proposed action requires that vessel operators and crews follow the PDCs described in Section C of Appendix A. These PDCs ensure all appropriate precautions are implemented to avoid collisions with ESA-listed species.

In addition, the PDCs described in Section B of Appendix A will help to ensure that these animals are extremely unlikely to be present during deployment of materials. . These PDCs require that deployments will be conducted only during daylight hours, when lighting, weather, and sea conditions allow for visual monitoring of the exclusion zone. Furthermore, NCDMF will follow the PDCs listed in Section B.5 of Appendix A, which contain an in-water construction moratorium for overwintering sturgeon on AR-130, AR-140, and AR-145 (all ocean reefs) from January 1 to March 31, and a September 30 to February 1 in-water work moratorium for AR-291, AR-392, and AR-396. AR-491 is an existing site that is also located in critical habitat; however, NCDMF and USACE have agreed that this site will not be included in the moratorium, as this site will be developed using only small rock or shell, similar to other oyster restoration activities. The deployment method of these types of material do not require construction materials that could be injurious to sturgeon, and are usually deployed by hand.

The siting of artificial reef material immediately adjacent of sea turtle nesting beaches may potentially increase the abundance of predatory species that may negatively impact offshore egress of sea turtle hatchlings. Hatchling sea turtles exhibit a "frenzy" period of swimming once in the water, whereupon they swim almost constantly in an offshore direction. For loggerhead sea turtles, this behavior likely serves to quickly extricate the hatchling from shallow coastal waters where predators may be more abundant (Wyneken and Salmon 1992) and disperse them into offshore waters that constitute their juvenile epipelagic habitat. Using the maximum documented swimming speed of 1.1808 ft/sec for hatchling loggerheads during their frenzy period (Salmon and Wyneken 1987; Witherington 1991) and based on an observed reduction of predation after 15 minutes of swimming from the beach regardless of the habitat type (i.e., sandy or reef) (Glenn 1996; Stewart and Wyneken 2004; Wyneken et al. 2000; Wyneken and Salmon 1992), it is anticipated that initial hatchling predation rates, on average, should be reduced approximately 1,100 ft (rounded up from 1,063 ft) off the nesting beach. Therefore, to avoid potential issues with increased predation of sea turtle hatchlings, PDC D.1 in Appendix A prohibits the deployment of artificial reef material within 1,100 ft of any nearshore area that predominantly consists of sand and is directly offshore of an identified nesting beach. Because artificial reef development is not expected to dramatically alter the predator complex in nearshore areas directly off nesting beaches that already have hard bottom or reef habitat in the immediate vicinity, a buffer would not be necessary in these areas. With the implementation of PDC D.1, we believe it is extremely unlikely that predation on hatchling sea turtles will occur due to the siting of artificial reef materials.

3.1.2 Effects related to the Temporary Loss of Habitat

ESA-listed sea turtles, fish, and marine mammals may be temporarily unable to use each reef site for foraging and shelter due to avoidance of reef deployment activities and related noise. We believe any effect will be insignificant because it will be temporary and of short duration (total duration of a typical deployment of materials is less than 1 week), intermittent (only occur during daylight hours), and will occur only a small area at any given time, relative to the sizes of adjacent areas of similar habitat.

3.1.3 Effects related to the Permanent Loss of Habitat

ESA-listed sturgeon feed on echinoderms, mollusks, and arthropods that can be found in open sand. Therefore, these species may be affected by the permanent loss of foraging habitat as a result of the placement of artificial reef material on open, sandy bottom. However, we believe that this effect will be insignificant given the extensive amount of similar habitat around each reef location. Sturgeon are opportunistic feeders and will be able to utilize the available habitat outside of the reef sites.

ESA-listed sea turtles, elasmobranchs (sharks and rays), and marine mammals may be present in the action area for resting, foraging, mating, and migration. We believe the effect of the permanent loss of open, sandy bottom will be insignificant on these species given the mobility of the species and available space around the artificial reefs to utilize for these activities. Furthermore, specifically with respect to foraging, oceanic whitetip sharks are a pelagic, highly-migratory and a high trophic-level predator that feed mainly on larger prey items that are present in the open ocean such as teleosts, cephalopods, marine mammals, other sharks and rays, crustaceans, and mollusks (Backus et al. 1956; Bonfil et al. 2009) Giant manta rays are filter feeders and primarily feed on surface zooplankton (Burgess et al. 2016; Couturier et al. 2013). Lastly, marine mammals are subsurface or surface filter feeders and are highly mobile. Therefore, we do not expect artificial reef structures that are placed on open, sandy bottom habitat to have any significant effects on feeding activities.

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

ESA-listed sea turtles may become entrapped (stuck) in an artificial reef structure. However, with the implementation of PDCs A.2, A.3, A.9, and A.10 in Appendix A, we believe this is extremely unlikely to occur and, therefore, discountable (with the exception of high-relief material for sea turtles). These PDCs require that steel reinforcement rods be cut from bridge spans at the base of the concrete to the extent possible and will not protrude more than 3 in. Reef structures, materials, and installation methods must also 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 in length. 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 in length if the third side is lowered to allow a 4 ft length opening edge on that third side (Figure 4). For instance, this would require a pyramid module with a 10 ft base that is 8 ft in height to 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. 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. 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.

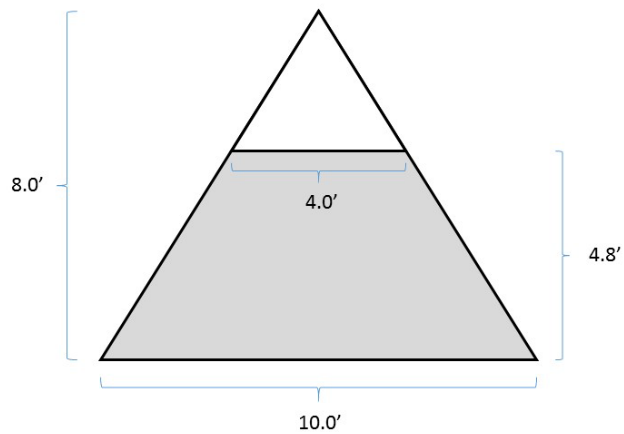

Figure 4. Standard Pyramid Module (i.e., 10 ft base and 8 ft height) with Minimum Required Measurements to Allow Adequate Sea Turtle Egress

It is extremely unlikely that blue, sei, sperm, and fin whales would be adversely affected by the proposed action. In the southeast U.S. Atlantic region, blue, sei, and sperm whales are predominantly found seaward of the continental shelf in deeper waters (Cetacean and Turtle Assessment Program 1982; NMFS 2011a; NMFS 2012b; Wenzel et al. 1988). Fin whales are generally found along the 100 m isobath with sightings also spread over deeper water including canyons along the shelf break (Waring et al. 2012). The artificial reef locations included in the proposed action either are outside of the primary range or depth of these species. In the unlikely event that one of these species goes into the action area, interactions with gear or reef structures are extremely unlikely to occur for the reasons stated below.

ESA-listed sea turtles, fish, and marine mammals may be physically injured or killed if they become entangled in abandoned fishing gear or other debris that may accumulate on low-relief and high-relief artificial reefs. We believe that any effects will be discountable for all ESA-listed species considered in this Opinion from low relief material and for ESA-listed fish and marine mammals from high relief materials for the following reasons. Low-relief and/or solid concrete material, rock rubble, and individual artificial reef modules present less complicated vertical relief that is not as likely to accumulate monofilament as larger, higher-relief materials, as documented in Barnette (2017). The implementation of the PDCs listed in Section A.4, 9, and 10 of Appendix A would further reduce the likelihood of entanglement. If these PDCs are implemented, the available information presented in Barnette (2017) indicates that gear and animal entanglement on low-relief material is extremely unlikely. With respect to high-relief artificial reef material, we anticipate that only sea turtles will experience entanglement events, and this is discussed further in Chapter 5. Entanglement of the ESA-listed fish or marine mammals that may be in the action area is extremely unlikely to occur. We have no information documenting any artificial reef entanglement event involving these species and it is also extremely unlikely that these species will utilize artificial reefs as habitat, thus decreasing any potential for interactions with accumulated monofilament. Alternatively, high-relief artificial reef material has been shown to have adverse effects on sea turtles, and is discussed further in Chapter 5.

ESA-listed sea turtles, fish, and marine mammals could also be injured or killed as a result of hooking or other interactions incidental to fishing activities in the vicinity of the proposed project; however, the potential for the proposed action to increase the risk of incidental capture is extremely unlikely, and any effect is therefore discountable. There is no evidence that establishment of artificial reefs increases the numbers of fishers or boats participating in a given fishery. Therefore, the relocation of fishing effort to new artificial reefs likely reduces fishing pressure at other locations, and may lessen the concentration of fishers by providing more locations at which to fish, reducing the likelihood of an ESA-listed species being hooked.

3.2. Analysis of Critical Habitat Not Likely to be Adversely Affected

The proposed action is located within the boundary of loggerhead sea turtle (NWA DPS) designated critical habitat (Constricted Migratory Habitat [LOGG-Unit N-01], Winter Habitat [Unit LOGG-N-02], and Nearshore Reproductive Habitat [Unit LOGG-N-03]), Atlantic sturgeon designated critical habitat (Carolina Unit 2, 3, and 4), and North Atlantic right whale designated critical habitat (Unit 2). We have determined that the proposed action is not likely to adversely affect any of this critical habitat.

3.2.1 Loggerhead Critical Habitat (NWA DPS)

The proposed action is located within the boundary of loggerhead sea turtle (NWA DPS) designated critical habitat (Constricted Migratory Habitat [LOGG-Unit N-01], Winter Habitat [LOGG-Unit N-02], and Nearshore Reproductive Habitat [LOGG-Unit N-03]).

Constricted Migratory Habitat is the high use migratory corridors that are constricted (limited in width) by land on one side and the edge of the continental shelf and Gulf Stream on the other side. The following primary constituent elements (PCEs) that support this habitat are as follows:

i. Constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways; and
ii. Passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas.

Winter habitat is the warm water habitat south of Cape Hatteras near the western edge of the Gulf Stream used by a high concentration of juveniles and adults during the winter months. The following PCEs that support this habitat are as follows:

i. Water temperatures above 10° C from November through April;
ii. Continental shelf waters in proximity to the western boundary of the Gulf Stream; and
iii. Water depths between 20 and 100 m.

Nearshore Reproductive Habitat is the portion of the nearshore waters adjacent to nesting beaches used by hatchlings to egress to the open-water environment as well as by nesting females to transit between beach and open water during the nesting season. The PCEs that support this habitat are as follows:

i. Nearshore waters directly off the highest density nesting beaches and their adjacent beaches, as identified in 50 CFR 17.95(c), to 1.6 kilometers (km) offshore;
ii. Waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water; and
iii. Waters with minimal manmade structures that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents.

The proposed action will have no effect on PCE (i) of the Constricted Migratory habitat, PCE (i) and (ii) of the Winter habitat, and PCE (i) of the Nearshore Reproductive habitat features.

The proposed action may affect PCE (ii) of the Constricted Migratory habitat feature by potentially creating an obstruction to sea turtle migration through the area. We believe this effect will be discountable due to the reef design and its depth. Since each reef has a minimum of a 10-ft vertical clearance, the existing and new artificial reefs are extremely unlikely to create an obstruction that would affect passage conditions allowing migration to and from nesting, breeding, and/or foraging areas.

The proposed action may affect PCE (iii) (water depths between 20 and 100 m) of Winter habitat by placing artificial reefs at depths within this range. We believe this effect will be insignificant because the artificial reef patch sizes are very small in comparison with the surrounding habitat available and have varying site depths (30 ft to 160 ft) offshore.

The proposed action may affect PCE (ii) of the Nearshore Reproductive habitat feature by potentially creating an obstruction to sea turtle transit through the area. We believe this effect will be discountable due to the reef design and depth. We believe the implementation of PDC D.1 in Appendix A would avoid disrupting the function of this critical habitat feature, as it would allow transit of nesting females and hatchlings. No artificial reef material will be deployed in any nearshore area 1,100 ft of any identified sea turtle nesting beach that predominantly consists of sandy benthic habitat. No emergent artificial reef material will be authorized in identified loggerhead sea turtle nearshore reproductive critical habitat areas. Any artificial reef material deployed within these critical habitat areas and within 1,100 ft of the beach at mean low water (MLW) must provide at least 4 ft of surface clearance at MLW, a maximum reef section length of 50 ft, and the project must include gaps free of any material at a 1:1 ratio (e.g., for every 25 ft of contiguous artificial reef material, a 25-ft gap clear of any material must be created). Lastly, the proposed action may affect PCE (iii) of the Nearshore Reproductive habitat by attracting more predators to the action area and therefore increasing predation upon hatchlings. We believe this effect will be discountable due to the implementation of the PDCs listed in Section D.1 of Appendix A (preventing the addition of artificial reefs within 1,100 ft of nesting beaches). As explained in more detail above, we anticipate that initial hatchling predation rates, on average, should be reduced by the time hatchlings swim this distance from the nesting beach (Barnette 2017).

3.2.2 *Atlantic Sturgeon Critical Habitat)*

Atlantic sturgeon critical habitat for the Carolina DPS includes those habitat components that support successful reproduction and recruitment. Four existing artificial reefs are located in Atlantic sturgeon critical habitat (AR-291, AR-392, AR-396, and AR-491). The Physical and Biological Features (PBFs) are as follows:

i. Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0-0.5 parts per thousand range) for settlement of fertilized eggs and refuge, growth, and development of early life stages;

ii. Aquatic habitat inclusive of waters with a gradual downstream gradient of 0.5 up to as high as 30 parts per thousand and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development;

iii. Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support:
   a. Unimpeded movement of adults to and from spawning sites;
   b. Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
   c. Staging, resting, or holding of subadults or spawning condition adults. Water depths in main river channels must also be deep enough (at least 1.2 meters [m]) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river;

iv. Water quality conditions, especially in the bottom meter of the water column, with temperature and oxygen values that support:
   a. Spawning;
   b. Annual and inter-annual adult, subadult, larval, and juvenile survival; and
   c. Larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and oxygen values will vary interdependently, and depending on salinity in a particular habitat.

The proposed action will have no effect on PBFs (i) and (iv).

Aquatic habitat inclusive of waters with a gradual downstream gradient PBF (ii) may be affected by the placement of artificial reef material. We believe the effect will be insignificant because this PBF refers to waters with a gradual downstream gradient of 0.5 to 30 parts per thousand and soft substrate between the river mouth and spawning sites for juvenile foraging and physiological development. The proposed action will not alter the downstream gradient. The placement of artificial reef materials will permanently remove available soft substrate that may be used for physiological development; however, there is abundant soft sandy substrate in and adjacent to the action area, and there are only 4 existing artificial reefs located in Atlantic sturgeon critical habitat and no additional reefs will be added in Atlantic sturgeon critical habitat as part of the proposed action evaluated under this Opinion.

The unobstructed water of appropriate depth PBF (iii) may be affected by the placement of reef material; however, we believe the effect will be insignificant. For the life stages potentially using the action area, PBF (iii) refers to water that is free from obstruction such that: spawning adults can make unencumbered movements to the spawning grounds; adults can access staging, resting, and holding habitats; and juveniles can make seasonal and physiologically-dependent movements to appropriate salinity zones within the river. The artificial reef sites are patchy and will have enough space between sites and patches that sturgeon can swim around the sites. PBF (iii) also refers to the requirement that the main channel be deep enough to ensure that there is continuous flow when adults or juveniles are there. There are only 4 existing reefs in critical habitat; therefore, the project will not alter the depth of the main channel or water flow in the channel. In addition, no new artificial reefs will be constructed in critical habitat, and the existing reefs currently located in critical habitat are only affecting a small portion of the action area and will not appreciably block sturgeon from accessing the mouth of the river.

3.2.3 North Atlantic Right Whale Critical Habitat

Unit 2 of North Atlantic right whale critical habitat is the portion off the coast of North Carolina, South Carolina, and Florida that provides calving areas. There are currently 7 existing artificial reefs located in North Atlantic right whale critical habitat (AR-420, AR-425, AR-430, AR-440, AR-445, AR-455, and AR-460). The PCEs are as follows:

1. Sea surface conditions associated with Force 4 or less on the Beaufort Scale
2. Sea surface temperatures of 7 degree Celsius (°C) to 17°C
3. Water depths of 6 to 28 meters, where these features simultaneously co-occur over contiguous areas of at least 231 nautical square miles (nmi^2) of ocean waters during the months of November through April.

The proposed action will have no effect on PCEs (i) and (ii).

We believe the only PCE that may be affected by the proposed action is PCE (iii). The proposed action may fragment large, contiguous areas where the essential features are present and alter the depth at certain sites and therefore may limit the availability of the essential features such that right whales are not able to select dynamic, optimal combinations of the features necessary for successful calving. We believe this effect will be insignificant. The proposed action will not occur at such a scale that there will be a measurable effect on water depth, making the area unsuitable to calving right whales. Also, the agreement to the PDCs listed in Section D.3 of Appendix A requires that no artificial reefs can be placed in water shallower than 30 ft deep, and allows only a maximum reef footprint of 1 nmi^2 in North Atlantic right whale critical habitat. If a new reef is added to an existing artificial reef, the total footprint of the combined reefs must not exceed 1 nmi^2. Also, the density of newly permitted reefs in North Atlantic right whale critical habitat cannot exceed 2 reefs (old or new) per 10 nmi^2.

3.3. Status of Species Likely to Be Adversely Affected

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

3.3.1 Sea Turtles

General Threats Faced by All Sea Turtle Species

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

Fisheries

Incidental bycatch in commercial fisheries is identified as a major contributor to past declines, and threat to future recovery, for all of the sea turtle species (NMFS et al. 2011a; NMFS and USFWS 1991; NMFS and USFWS 1992; NMFS and USFWS 1993; NMFS and USFWS 2008). Domestic fisheries often capture, injure, and kill sea turtles at various life stages. Sea turtles in the pelagic environment are exposed to U.S. Atlantic pelagic longline fisheries. Sea turtles in the benthic environment in waters off the coastal U.S. are exposed to a suite of other fisheries in federal and state waters. These fishing methods include trawls, gillnets, purse seines, hook-and-line gear (including bottom longlines and vertical lines [e.g., bandit gear, handlines, and rod-reel]), pound nets, and trap fisheries. Refer to the Environmental Baseline section of this Opinion for more specific information regarding federal and state managed fisheries affecting sea turtles within the action area). The Southeast U.S. shrimp fisheries have historically been the largest fishery threat to benthic sea turtles in the southeastern U.S., 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 U.S., 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 1997a). Sea turtles entering coastal or inshore areas have also been affected by entrainment in the cooling-water systems of electrical generating plants. Other nearshore threats include harassment and/or injury resulting from private and commercial vessel operations, military detonations and training exercises, in-water construction activities, and scientific research activities.

Coastal Development and Erosion Control

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

Environmental Contamination

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

The April 20, 2010, explosion of the Deepwater Horizon (DWH) oil rig affected sea turtles in the Gulf of Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (DWH Trustees 2015). Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in Sargassum algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. The spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. Information on the spill impacts to individual sea turtle species is presented in the Status of the Species sections for each species.

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

Climate Change

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

Climate change impacts on sea turtles currently cannot be predicted with any degree of certainty; however, significant impacts to the hatchling sex ratios of sea turtles may result (NMFS and USFWS 2007c). 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 2007c).

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 2007e). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005). The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

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

Other Threats

Predation by various land predators is a threat to developing nests and emerging hatchlings. The major natural predators of sea turtle nests are mammals, including raccoons, dogs, pigs, skunks, and badgers. Emergent hatchlings are preyed upon by these mammals as well as ghost crabs, laughing gulls, and the exotic South American fire ant (*Solenopsis invicta*). In addition to natural predation, direct harvest of eggs and adults from beaches in foreign countries continues to be a problem for various sea turtle species throughout their ranges (NMFS and USFWS 2008). Diseases, toxic blooms from algae and other microorganisms, and cold stunning events are additional sources of mortality that can range from local and limited to wide-scale and impacting hundreds or thousands of animals.

3.3.1.1 Loggerhead Sea Turtle (NWA DPS)

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. NMFS and USFWS published a Final Rule which designated 9 DPSs for loggerhead sea turtles (76 Federal Register [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 centimeters [cm]) long, measured as a straight carapace length (SCL), and weigh approximately 255 pounds (lb) (116 kilograms [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 (National Research Council 1990a). For the NWA DPS, most nesting occurs along the coast of the United States, from southern Virginia to Alabama. Additional nesting beaches for this DPS are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba (Moncada-Gavilán 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

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

Within the NWA DPS, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf Coast of Florida. Previous Section 7 analyses have recognized at least 5 western

Atlantic subpopulations, divided geographically as follows: (1) a Northern nesting subpopulation, occurring from North Carolina to northeast Florida at about 29ºN; (2) a South Florida nesting subpopulation, occurring from 29°N on the east coast of the state to Sarasota on the west coast; (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez M. 1990; Turtle Expert Working Group 2000); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS 2001).

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles concluded that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida Peninsula. It also concluded that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the recovery plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are as follows: (1) the Northern Recovery Unit (Florida/Georgia border north through southern Virginia), (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The recovery plan concluded that all recovery units are essential to the recovery of the species. Although the recovery plan was written prior to the listing of the NWA DPS, the recovery units for what was then termed the Northwest Atlantic population apply to the NWA DPS.

Life History Information

The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, which include the ecosystems those stages generally use: (1) egg (terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone[3]), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting female (terrestrial zone) (NMFS and USFWS 2008). Loggerheads are long-lived animals. They reach sexual maturity between 20-38 years of age, although age of maturity varies widely among populations (Frazer and Ehrhart 1985; NMFS 2001). The annual mating season occurs from late March to early June, and female turtles lay eggs throughout the summer months. Females deposit an average of 4.1 nests within a nesting season (Murphy and Hopkins 1984), but an individual female only nests every 3.7 years on average (Tucker 2010). Each nest contains an average of 100-126 eggs (Dodd Jr. 1988) which incubate for 42-75 days before hatching (NMFS and USFWS 2008). Loggerhead hatchlings are 1.5-2 in long and weigh about 0.7 ounces (oz) (20 g).

As post-hatchlings, loggerheads hatched on U.S. beaches enter the “oceanic juvenile” life stage, migrating offshore and becoming associated with *Sargassum* habitats, driftlines, and other convergence zones (Carr 1986; Conant et al. 2009; Witherington 2002). Oceanic juveniles grow at rates of 1-2 in (2.9-5.4 cm) per year (Bjorndal et al. 2003; Snover 2002) over a period as long

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

as 7-12 years (Bolten et al. 1998) before moving to more coastal habitats. Studies have suggested that not all loggerhead sea turtles follow the model of circumnavigating the North Atlantic Gyre as pelagic juveniles, followed by permanent settlement into benthic environments (Bolten and Witherington 2003; Laurent et al. 1998). These studies suggest some turtles may either remain in the oceanic habitat in the North Atlantic longer than hypothesized, or they move back and forth between oceanic and coastal habitats interchangeably (Witzell 2002). Stranding records indicate that when immature loggerheads reach 15-24 in (40-60 cm) SCL, they begin to reside in coastal inshore waters of the continental shelf throughout the U.S. Atlantic and Gulf of Mexico (Witzell 2002).

After departing the oceanic zone, neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Estuarine waters of the United States, including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, Florida Bay, as well as numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads (Conant et al. 2009).

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

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

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

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

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

In addition to the total nest count estimates, FWRI uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. This provides a better tool for understanding the nesting trends (Figure 5). FWRI performed a detailed analysis of the long-term loggerhead index nesting data (1989-2017; http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). Over that time period, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that was followed by a sharp decline over the subsequent 9 years. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represented a new record for loggerheads on the core index beaches. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but nonsignificant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening confidence intervals (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). Nesting at the core index beaches declined in 2017 to 48,033, which is still the 4th highest total since 2001.

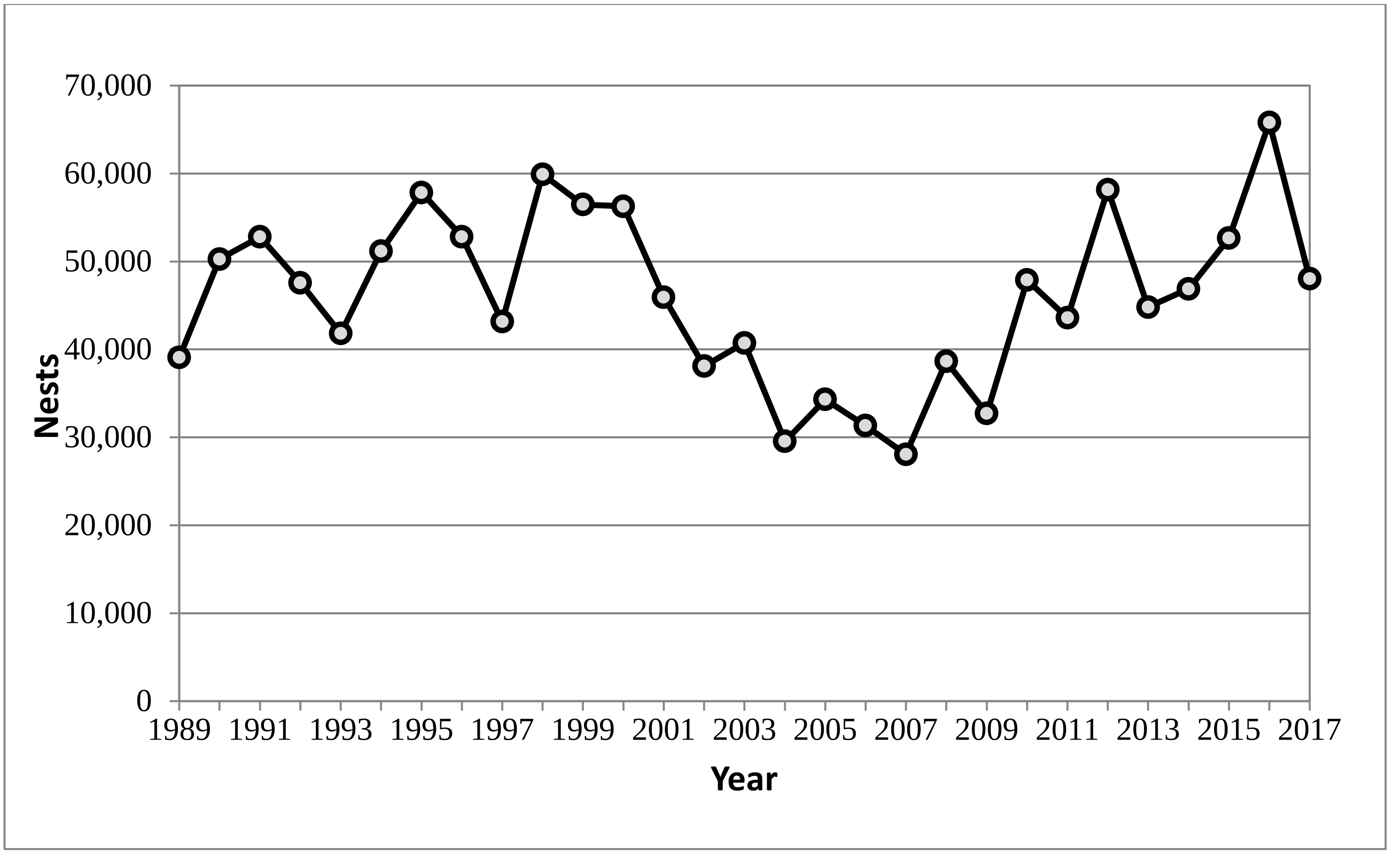

Figure 5. Loggerhead Sea Turtle Nesting at Florida Index Beaches Since 1989

Northern Recovery Unit

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

Data since that analysis (Table 4) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, GADNR press release, http://www.georgiawildlife.com/node/3139). South Carolina and North Carolina nesting have also begun to shift away from the past declining trend. Loggerhead nesting in Georgia, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016.

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

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

South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record (Figure 6).

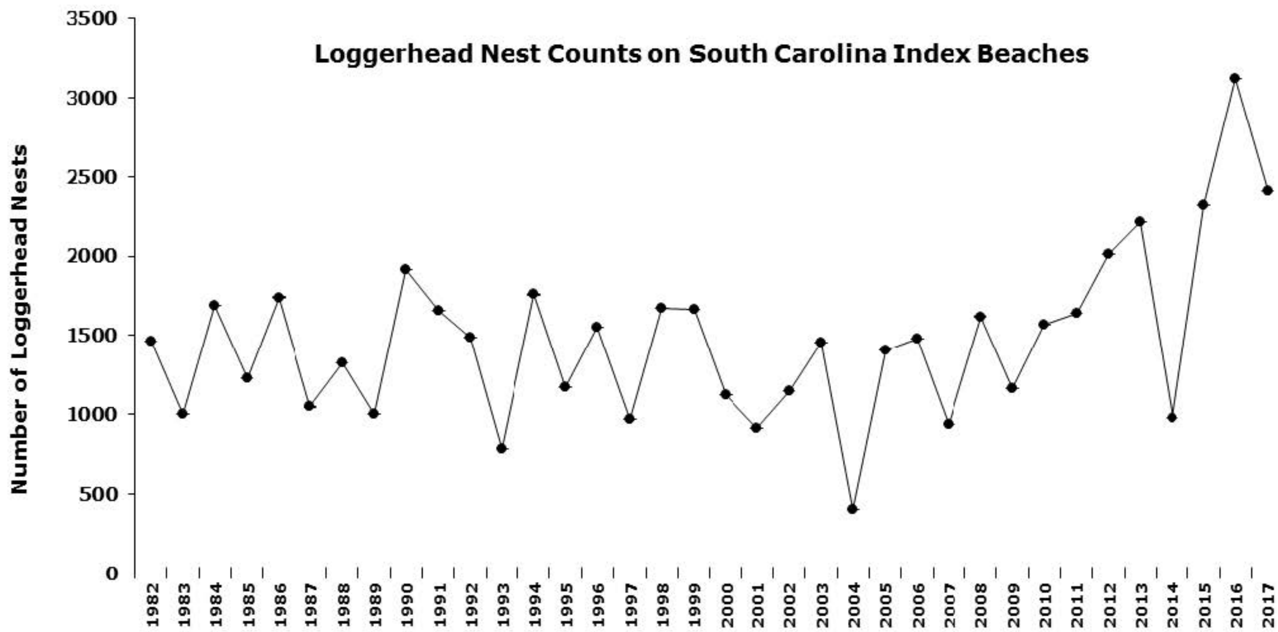

Figure 6. South Carolina Index Nesting Beach Counts for Loggerhead Sea Turtles (from the SCDNR website: http://www.dnr.sc.gov/seaturtle/nest.htm)

Other Northwest Atlantic DPS Recovery Units

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

In-water Trends

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

Population Estimate

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

Threats (Specific to Loggerhead Sea Turtles)
The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 3.3.1. Yet the impact of fishery interactions is a point of further emphasis for this species. The joint NMFS and USFWS Loggerhead Biological Review Team determined that the greatest threats to the NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant et al. 2009).

Regarding the impacts of pollution, loggerheads may be particularly affected by organochlorine contaminants; they have the highest organochlorine concentrations (Storelli et al. 2008) and metal loads (D'Ilio et al. 2011) in sampled tissues among the sea turtle species. It is thought that dietary preferences were likely to be the main differentiating factor among sea turtle species. Storelli et al. (2008) analyzed tissues from stranded loggerhead sea turtles and found that mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991).

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the Deepwater Horizon (DWH) oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

Unlike Kemp's ridleys, the majority of nesting for the Northwest Atlantic Ocean loggerhead DPS occurs on the Atlantic coast, and thus loggerheads were impacted to a relatively lesser degree. However, it is likely that impacts to the NGMRU of the NWA loggerhead 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 Trustees estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH oil spill response activities on nesting beaches. Although the long-term effects remain unknown, the DWH oil spill event impacts to the Northern Gulf of Mexico Recovery Unit may result in some nesting declines in the future due to a large reduction of oceanic age classes during the DWH oil spill event. Although adverse impacts occurred to loggerheads, the proportion of the population that is expected to have been exposed to and directly impacted by the DWH oil spill event is relatively low. Thus we do not believe a population-level impact occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

Specific information regarding potential climate change impacts on loggerheads is also available. Modeling suggests an increase of 2°C in air temperature would result in a sex ratio of over 80% female offspring for loggerheads nesting near Southport, North Carolina. The same increase in air temperatures at nesting beaches in Cape Canaveral, Florida, would result in close to 100% female offspring. Such highly skewed sex ratios could undermine the reproductive capacity of the species. More ominously, an air temperature increase of 3°C is likely to exceed the thermal threshold of most nests, leading to egg mortality (Hawkes et al. 2007). Warmer sea surface temperatures have also been correlated with an earlier onset of loggerhead nesting in the spring (Hawkes et al. 2007; Weishampel et al. 2004), short inter-nesting intervals (Hays et al. 2002), and shorter nesting seasons (Pike et al. 2006).

3.3.1.2 Green Sea Turtle (NA and SA DPSs)

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 distinct population segments (DPSs) (81 FR 20057) (Figure 7). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific DPSs were listed as threatened. For the purposes of this consultation, only the SA DPS and NA DPS will be considered, as they are the only two DPSs with individuals occurring in the Atlantic and Gulf of Mexico waters of the United States.

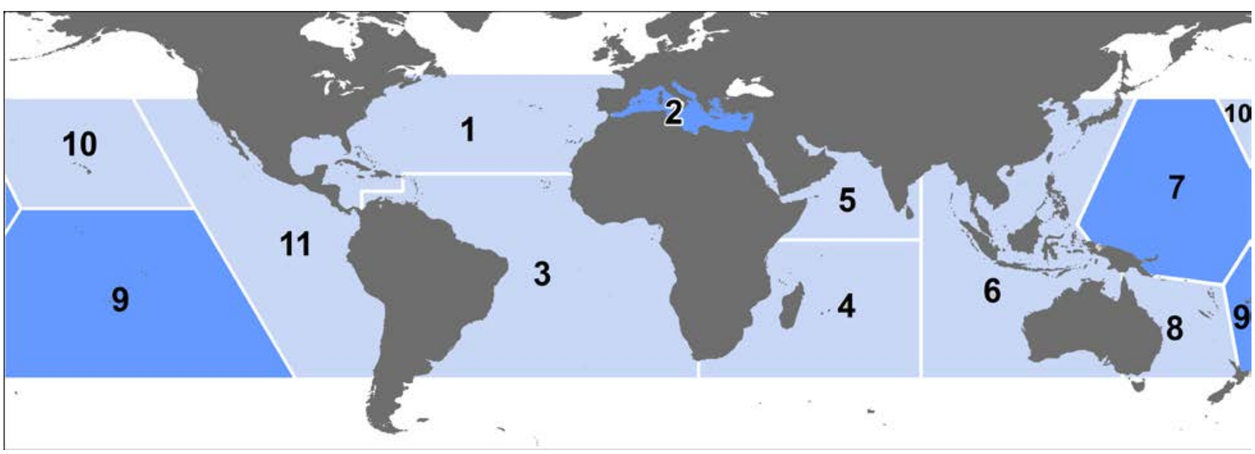

Figure 7. Threatened (light) and Endangered (dark) Green Turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11.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

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

With the exception of post-hatchlings, green sea turtles live in nearshore tropical and subtropical waters where they generally feed on marine algae and seagrasses. They have specific foraging grounds and may make large migrations between these forage sites and natal beaches for nesting (Hays et al. 2001). Green sea turtles nest on sandy beaches of mainland shores, barrier islands, coral islands, and volcanic islands in more than 80 countries worldwide (Hirth 1997). The 2 largest nesting populations are found at Tortuguero, on the Caribbean coast of Costa Rica (part of the NA DPS), and Raine Island, on the Pacific coast of Australia along the Great Barrier Reef.

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

<u>North Atlantic DPS Distribution</u>

The NA DPS boundary is illustrated in Figure 7. Four regions support nesting concentrations of particular interest in the NA DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto

Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

The complete nesting range of NA DPS green sea turtles within the southeastern United States includes sandy beaches between Texas and North Carolina, as well as Puerto Rico (NMFS and USFWS 1991; Piniak et al. 2007). The vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Principal U.S. nesting areas for green sea turtles are in eastern Florida, predominantly Brevard south through Broward counties.

In U.S. Atlantic and Gulf of Mexico waters, green sea turtles are distributed throughout inshore and nearshore waters from Texas to Massachusetts. Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984; Hildebrand 1982; Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1991; Wershoven and Wershoven 1992). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatán Peninsula.

South Atlantic DPS Distribution

The SA DPS boundary is shown in Figure 7, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

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

Life History Information

Green sea turtles reproduce sexually, and mating occurs in the waters off nesting beaches and along migratory routes. Mature females return to their natal beaches (i.e., the same beaches where they were born) to lay eggs (Balazs 1979; Frazer and Ehrhart 1985) every 2-4 years while males are known to reproduce every year (Balazs 1983). In the southeastern United States, females generally nest between June and September, and peak nesting occurs in June and July (Witherington and Ehrhart 1989b). During the nesting season, females nest at approximately 2-week intervals, laying an average of 3-4 clutches (Johnson and Ehrhart 1996). Clutch size often varies among subpopulations, but mean clutch size is approximately 110-115 eggs. In Florida, green sea turtle nests contain an average of 136 eggs (Witherington and Ehrhart 1989b). Eggs incubate for approximately 2 months before hatching. Hatchling green sea turtles are approximately 2 in (5 cm) in length and weigh approximately 0.9 oz (25 grams [g]). Survivorship at any particular nesting site is greatly influenced by the level of man-made stressors, with the more pristine and less disturbed nesting sites (e.g., along the Great Barrier Reef in Australia) showing higher survivorship values than nesting sites known to be highly disturbed (e.g., Nicaragua) (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 2007a). Green sea turtles exhibit particularly slow growth rates of about 0.4-2 in (1-5 cm) per year (Green 1993), which may be attributed to their largely herbivorous, low-net energy diet (Bjorndal 1982). At approximately 8-10 in (20-25 cm) carapace length, juveniles leave the pelagic environment and enter nearshore developmental habitats such as protected lagoons and open coastal areas rich in sea grass and marine algae. Growth studies using skeletochronology indicate that green sea turtles in the western Atlantic shift from the oceanic phase to nearshore developmental habitats after approximately 5-6 years (Bresette et al. 2006; Zug and Glor 1998). Within the developmental habitats, juveniles begin the switch to a more herbivorous diet, and by adulthood feed almost exclusively on seagrasses and algae (Ingle and Smith 1974), although some populations are known to also feed heavily on invertebrates (Carballo et al. 2002). Green sea turtles mature slowly, requiring 20-50 years to reach sexual maturity (Chaloupka and Musick 1997; Hirth 1997).

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

Status and Population Dynamics

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

North Atlantic DPS

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

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 2007a). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually.

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

In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 8). According to data collected from Florida's index nesting beach survey from 1989-2017, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 38,954 in 2017. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011, and a return to the trend of biennial peaks in abundance thereafter (Figure 8). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years.

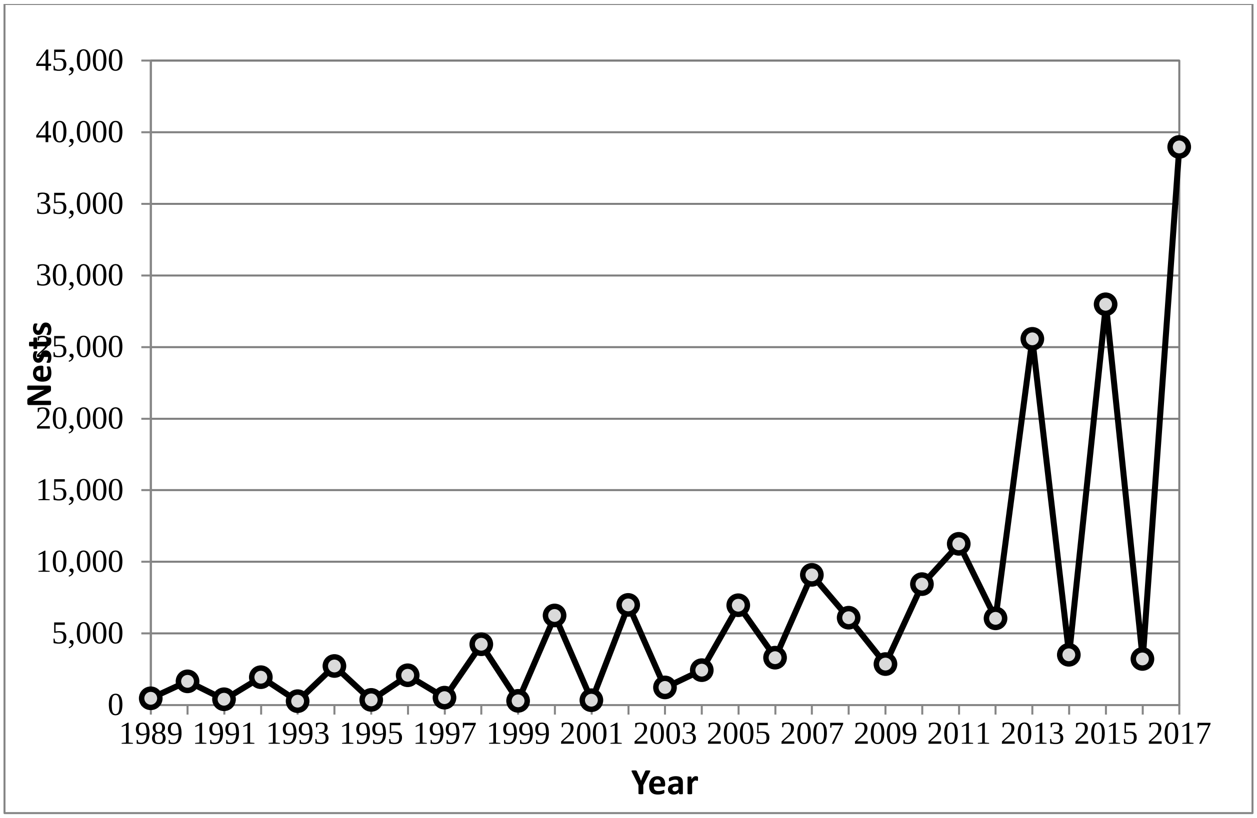

Figure 8. Green Sea Turtle Nesting at Florida Index Beaches Since 1989

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

South Atlantic DPS

The SA DPS is large, estimated at over 63,000 nesters, but data availability is poor. More than half of the 51 identified nesting sites (37) did not have sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). This includes some sites, such as beaches in French Guiana, which are suspected to have large numbers of nesters. Therefore, while the estimated number of nesters may be substantially underestimated, we also do not know the population trends at those data-poor beaches. However, while the lack of data was a concern due to increased uncertainty, the overall trend of the SA DPS was not considered to be a major concern as some of the largest nesting beaches such as Ascension Island, Aves Island (Venezuela), and Galibi (Suriname) appear to be increasing. Others such as Trindade (Brazil), Atol das Rocas (Brazil), and Poilão and the rest of Guinea-Bissau seem to be stable or do not have sufficient data to make a determination. Bioko (Equatorial Guinea) appears to be in decline but has less nesting than the other primary sites (Seminoff et al. 2015).

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

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.3.1.

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

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

Whereas oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juveniles greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources, which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

While green turtles regularly use the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic, and the proportion of the population using the northern Gulf of Mexico at any given time is relatively low. Although it is known that adverse impacts occurred and numbers of animals in the Gulf of Mexico were reduced as a result of the DWH, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event, as well as the impacts being primarily to smaller juveniles (lower reproductive value than adults and large juveniles), reduces the impact to the overall population. It is unclear what impact these losses may have caused on a population level, but it is not expected to have had a large impact on the population trajectory moving forward. However, recovery of green turtle numbers equivalent to what was lost in the northern Gulf of Mexico as a result of the spill will likely take decades of sustained efforts to reduce the existing threats and enhance survivorship of multiple life stages (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016).

3.3.1.3 Kemp's ridley Sea Turtle

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

Species Description and Distribution
The Kemp's ridley sea turtle is the smallest of all sea turtles. Adults generally weigh less than 100 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 millimeters [mm]) 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 1989a), although the time spent in the oceanic zone may vary from 1-4 years or perhaps more (Turtle Expert Working Group 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 9), which indicates the species is recovering.

It is worth noting that when the Bi-National Kemp's Ridley Sea Turtle Population Restoration Project was initiated in 1978, only Rancho Nuevo nests were recorded. In 1988, nesting data from southern beaches at Playa Dos and Barra del Tordo were added. In 1989, data from the northern beaches of Barra Ostionales and Tepehuajes were added, and most recently in 1996, data from La Pesca and Altamira beaches were recorded. Currently, nesting at Rancho Nuevo accounts for just over 81% of all recorded Kemp's ridley nests in Mexico. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley nests in Mexico increased to 21,797 in 2012 (Burchfield 2013). From 2013 through 2014, there was a second significant decline, as only 16,385 and 11,279 nests were recorded, respectively. More recent data, however, indicated an increase in nesting. In 2015 there were 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Burchfield 2013). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm., August 31, 2017), but nesting for 2018 has declined to 17,945 (Gladys Porter Zoo data presentation by J. Pena, 2018). At this time, it is unclear whether the increases and declines in nesting seen over the past decade represents a population oscillating around an equilibrium point or if nesting will decline or increase in the future.

A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (National Park Service data, http://www.nps.gov/pais/naturescience/strp.htm, http://www.nps.gov/pais/naturescience/current-season.htm). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015.

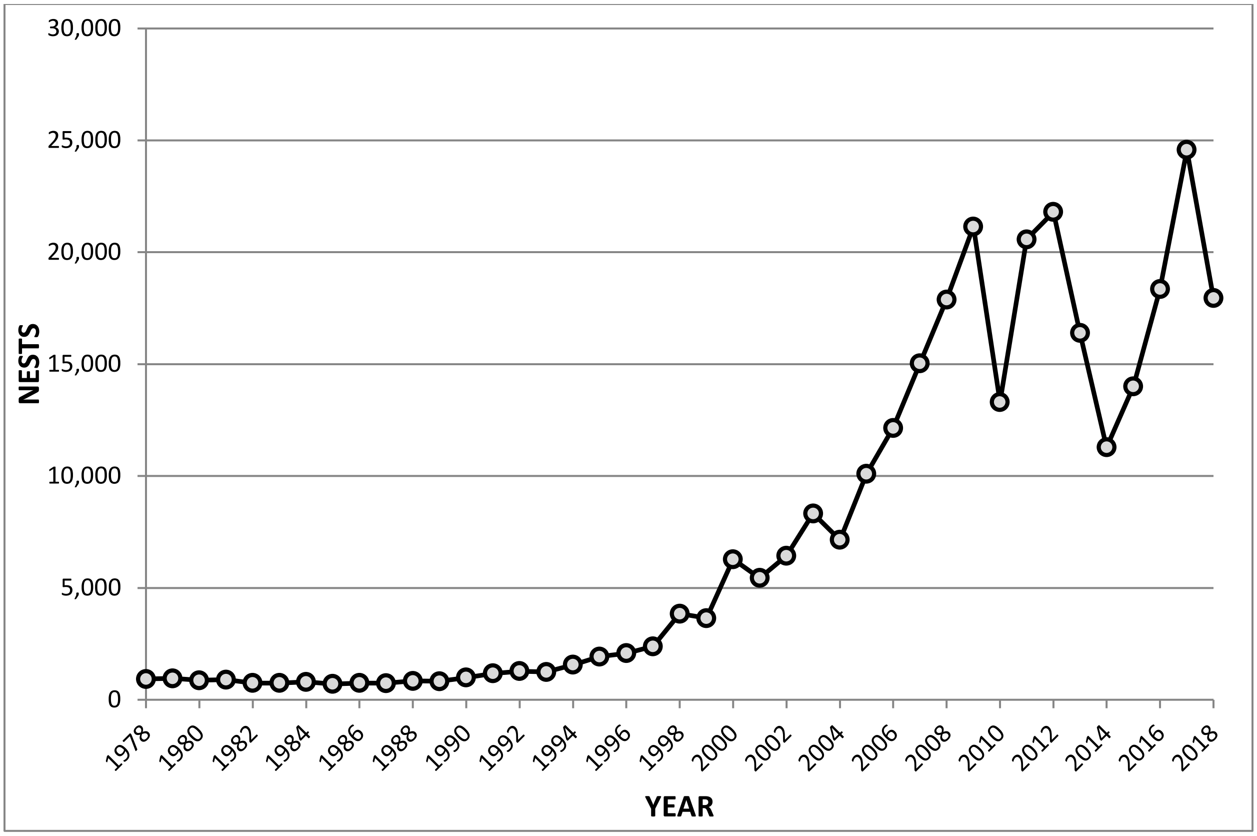

Figure 9. Kemp's ridley Nest Totals from Mexican Beaches (Gladys Porter Zoo nesting database 2017)

Through modelling, Heppell et al. (2005) predicted the population is expected to increase at least 12-16% per year and could reach at least 10,000 females nesting on Mexico beaches by 2015. NMFS et al. (2011) produced an updated model that predicted the population to increase 19% per year and to attain at least 10,000 females nesting on Mexico beaches by 2011. Approximately 25,000 nests would be needed for an estimate of 10,000 nesters on the beach, based on an average 2.5 nests/nesting female. While counts did not reach 25,000 nests by 2015, it is clear that the population has increased over the long term. The increases in Kemp's ridley sea turtle nesting over the last 2 decades is likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of TEDs, reduced trawling effort in Mexico and the United States, and possibly other changes in vital rates (Turtle Expert Working Group 1998; Turtle Expert Working Group 2000). While these results are encouraging, the species' limited range as well as low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental randomness, all factors which are often difficult to predict with any certainty. Additionally, the significant nesting declines observed in 2010 and 2013-2014 potentially indicate a serious population-level impact, and there is cause for concern regarding the ongoing recovery trajectory.

Threats

Kemp's ridley sea turtles face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.3.1; 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[4] 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.

Over the past 6 years, NMFS has documented (via the Sea Turtle Stranding and Salvage Network data, http://www.sefsc.noaa.gov/species/turtles/strandings.htm) elevated sea turtle strandings in the Northern Gulf of Mexico, particularly throughout the Mississippi Sound area. In the first 3 weeks of June 2010, over 120 sea turtle strandings were reported from Mississippi and Alabama waters, none of which exhibited any signs of external oiling to indicate effects associated with the DWH oil spill event. A total of 644 sea turtle strandings were reported in 2010 from Louisiana, Mississippi, and Alabama waters, 561 (87%) of which were Kemp's ridley sea turtles. During March through May of 2011, 267 sea turtle strandings were reported from Mississippi and Alabama waters alone. A total of 525 sea turtle strandings were reported in 2011 from Louisiana, Mississippi, and Alabama waters, with the majority (455) having occurred from March through July, 390 (86%) of which were Kemp's ridley sea turtles. During 2012, a total of 384 sea turtles were reported from Louisiana, Mississippi, and Alabama waters. Of these reported strandings, 343 (89%) were Kemp's ridley sea turtles. During 2014, a total of 285 sea turtles were reported from Louisiana, Mississippi, and Alabama waters, though the data is incomplete. Of these reported strandings, 229 (80%) were Kemp's ridley sea turtles. These stranding numbers are significantly greater than reported in past years; Louisiana, Mississippi, and Alabama waters reported 42 and 73 sea turtle strandings for 2008 and 2009, respectively. It should be noted that stranding coverage has increased considerably due to the DWH oil spill event.

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

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

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 fishery during the summer of 2012. During May-July of that year, observers reported 24 sea turtle interactions in the skimmer trawl fishery. All but a single sea turtle were identified as Kemp's ridleys (1 sea turtle was an unidentified hardshell turtle). Encountered sea turtles were all very small juvenile specimens, ranging from 7.6-19.0 in (19.4-48.3 cm) curved carapace length (CCL). All sea turtles were released alive. The small average size of encountered Kemp's ridleys introduces a potential conservation issue, as over 50% of these reported sea turtles could potentially pass through the maximum 4-in bar spacing of TEDs currently required in the shrimp fishery. Due to this issue, a proposed 2012 rule to require TEDs in the skimmer trawl fishery (77 FR 27411) was not implemented. Based on anecdotal information, these interactions were a relatively new issue for the inshore skimmer trawl fishery. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of Mexico may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles are considered here. Kemp's ridleys experienced the greatest negative impact stemming from the DWH oil spill event of any sea turtle species. Impacts to Kemp's ridley sea turtles occurred to offshore small juveniles, as well as large juveniles and adults. Loss of hatchling production resulting from injury to adult turtles was also estimated for this species. Injuries to adult turtles of other species, such as loggerheads, certainly would have resulted in unrealized nests and hatchlings to those species as well. Yet, the calculation of unrealized nests and hatchlings was limited to Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of Mexico throughout their lives (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016).

A total of 217,000 small juvenile Kemp's ridleys (51.5% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. That means approximately half of all small juvenile Kemp's ridleys from the total population estimate of 430,000 oceanic small juveniles were exposed to oil. Furthermore, a large number of small juveniles were removed from the population, as up to 90,300 small juveniles Kemp's ridleys are estimated to have died as a direct result of the exposure. Therefore, as much as 20% of the small oceanic juveniles of this species were killed during that year. Impacts to large juveniles (>3 years old) and adults were also high. An estimated 21,990 such individuals were exposed to oil (about 22% of the total estimated population for those age classes); of those, 3,110 mortalities were estimated (or 3% of the population for those age classes). The loss of near-reproductive and reproductive-stage females would have contributed to some extent to the decline in total nesting abundance observed between 2011 and 2014. The estimated number of unrealized Kemp's ridley nests is between 1,300 and 2,000, which translates to between approximately

65,000 and 95,000 unrealized hatchlings (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). This is a minimum estimate, however, because the sublethal effects of the DWH oil spill event on turtles, their prey, and their habitats might have delayed or reduced reproduction in subsequent years, which may have contributed substantially to additional nesting deficits observed following the DWH oil spill event. These sublethal effects could have slowed growth and maturation rates, increased remigration intervals, and decreased clutch frequency (number of nests per female per nesting season). The nature of the DWH oil spill event effect on reduced Kemp's ridley nesting abundance and associated hatchling production after 2010 requires further evaluation. It is clear that the DWH oil spill event resulted in large losses to the Kemp's ridley population across various age classes, and likely had an important population-level effect on the species. Still, we do not have a clear understanding of those impacts on the population trajectory for the species into the future.

3.3.1.4 Leatherback Sea Turtle

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

Species Description and Distribution

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

Unlike other sea turtles, leatherbacks have several unique traits that enable them to live in cold water. For example, leatherbacks have a countercurrent circulatory system (Greer Jr. et al. 1973),[5] a thick layer of insulating fat (Davenport et al. 1990; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),[6] and they can increase their body temperature through increased metabolic activity (Bostrom and Jones 2007; Southwood et al. 2005). These adaptations allow leatherbacks to be comfortable in a wide range of temperatures, which helps them to travel further than any other sea turtle species (NMFS and USFWS 1995). For example, a leatherback may swim more than 6,000 miles (mi) (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).

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

[6] "Gigantothermy" refers to a condition when an animal has relatively high volume compared to its surface area, and as a result, it loses less heat.

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. 1989a), but they may also come into shallow waters to locate prey items.

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

Life History Information

The leatherback life cycle is broken into several stages: (1) egg/hatchling, (2) post-hatchling, (3) juvenile, (4) subadult, and (5) adult. Leatherbacks are a long-lived species that delay age of maturity, have low and variable survival in the egg and juvenile stages, and have relatively high and constant annual survival in the subadult and adult life stages (Chaloupka 2002; Crouse 1999; Heppell et al. 1999; Heppell et al. 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 (García-Muñoz and Sarti 2000; McDonald and Dutton 1996; Spotila et al. 2000). Unlike other sea turtle species, female leatherbacks do not always nest at the same beach year after year; some females may even nest at different beaches during the same year (Dutton et al. 2005; Eckert et al. 1989b; Keinath and Musick 1993; Steyermark et al. 1996). Individual female leatherbacks have been observed with fertility spans as long as 25 years (Hughes 1996). Females usually lay up to 10 nests during the 3-6 month nesting season (March through July in the United States), typically 8-12 days apart, with 100 eggs or more per nest (Eckert et al. 2012; Eckert et al. 1989b; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). Yet, up to approximately 30% of the eggs may be infertile (Eckert et al. 1989b; Eckert et al. 1984; Maharaj 2004; Matos 1986;

Stewart and Johnson 2006; Tucker 1988). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50% worldwide (Eckert et al. 2012), which is lower than the greater than 80% reported for other sea turtle species (Miller 1997). In the United States, the emergent success is higher at 54-72% (Eckert and Eckert 1990; Stewart and Johnson 2006; Tucker 1988). Thus the number of hatchlings in a given year may be less than the total number of eggs produced in a season. Eggs hatch after 60-65 days, and the hatchlings have white striping along the ridges of their backs and on the edges of the flippers. Leatherback hatchlings weigh approximately 1.5-2 oz (40-50 g), and have lengths of approximately 2-3 in (51-76 mm), with fore flippers as long as their bodies. Hatchlings grow rapidly, with reported growth rates for leatherbacks from 2.5-27.6 in (6-70 cm) in length, estimated at 12.6 in (32 cm) per year (Jones et al. 2011).

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

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

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

Status and Population Dynamics

The status of the Atlantic leatherback population has been less clear than the Pacific population, which has shown dramatic declines at many nesting sites (Martínez et al. 2007; Santidrián Tomillo et al. 2007; Spotila et al. 2000). This uncertainty has been a result of inconsistent beach and aerial surveys, cycles of erosion, and reformation of nesting beaches in the Guianas (representing the largest nesting area). Leatherbacks also show a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species. Coordinated efforts of data collection and analyses by the leatherback Turtle Expert Working Group have helped to clarify the understanding of the Atlantic population status (Turtle Expert Working Group 2007).

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (Turtle Expert Working Group 2007). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with most of the nesting occurring in the Guianas and Trinidad. The Southern Caribbean/Guianas stock of leatherbacks was designated after genetics studies indicated that animals from the Guianas (and possibly Trinidad) should be viewed as a single population. Using nesting females as a proxy for population, the Turtle Expert Working Group (2007) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate. TEWG observed positive growth within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (Turtle Expert Working Group 2007). More specifically, Tiwari et al. (2013) report an estimated three-generation abundance change of +3%, +20,800%, +1,778%, and +6% in Trinidad, Guyana, Suriname, and French Guiana, respectively.

Researchers believe the cyclical pattern of beach erosion and then reformation has affected leatherback nesting patterns in the Guianas. For example, between 1979 and 1986, the number of leatherback nests in French Guiana had increased by about 15% annually (NMFS 2001). This increase was then followed by a nesting decline of about 15% annually. This decline corresponded with the erosion of beaches in French Guiana and increased nesting in Suriname. This pattern suggests that the declines observed since 1987 might actually be a part of a nesting cycle that coincides with cyclic beach erosion in Guiana (Schulz 1975). Researchers think that the cycle of erosion and reformation of beaches may have changed where leatherbacks nest throughout this region. The idea of shifting nesting beach locations was supported by increased nesting in Suriname,[7] while the number of nests was declining at beaches in Guiana (Hilterman et al. 2003). Though this information suggested the long-term trend for the overall Suriname and French Guiana population was increasing.

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. Across the Western Caribbean, nesting is most prevalent in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coastline of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from index nesting beaches in Tortuguero, Gandoca, and Pacuaré in Costa Rica indicate that the nesting population likely was not growing over the 1995-2005 time series (Turtle Expert Working Group 2007). 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.

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

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

between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (Turtle Expert Working Group 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 (Turtle Expert Working Group 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 (Turtle Expert Working Group 2007).

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 Turtle Expert Working Group (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. Florida Fish and Wildlife Conservation Commission (FWC) Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 10 and Table 5). A similar pattern was also observed statewide (Table 5). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting. Overall, the trend showed growth on Florida's east coast beaches. Tiwari et al. (2013) report an annual growth rate of 9.7% and a three-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, with a partial rebound in 2018. Similar patterns are being seen in other nesting beaches of the NW Atlantic. A status review is currently underway to analyze leatherback status and trends worldwide.

Table 5. Number of Leatherback Sea Turtle Nests in Florida

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

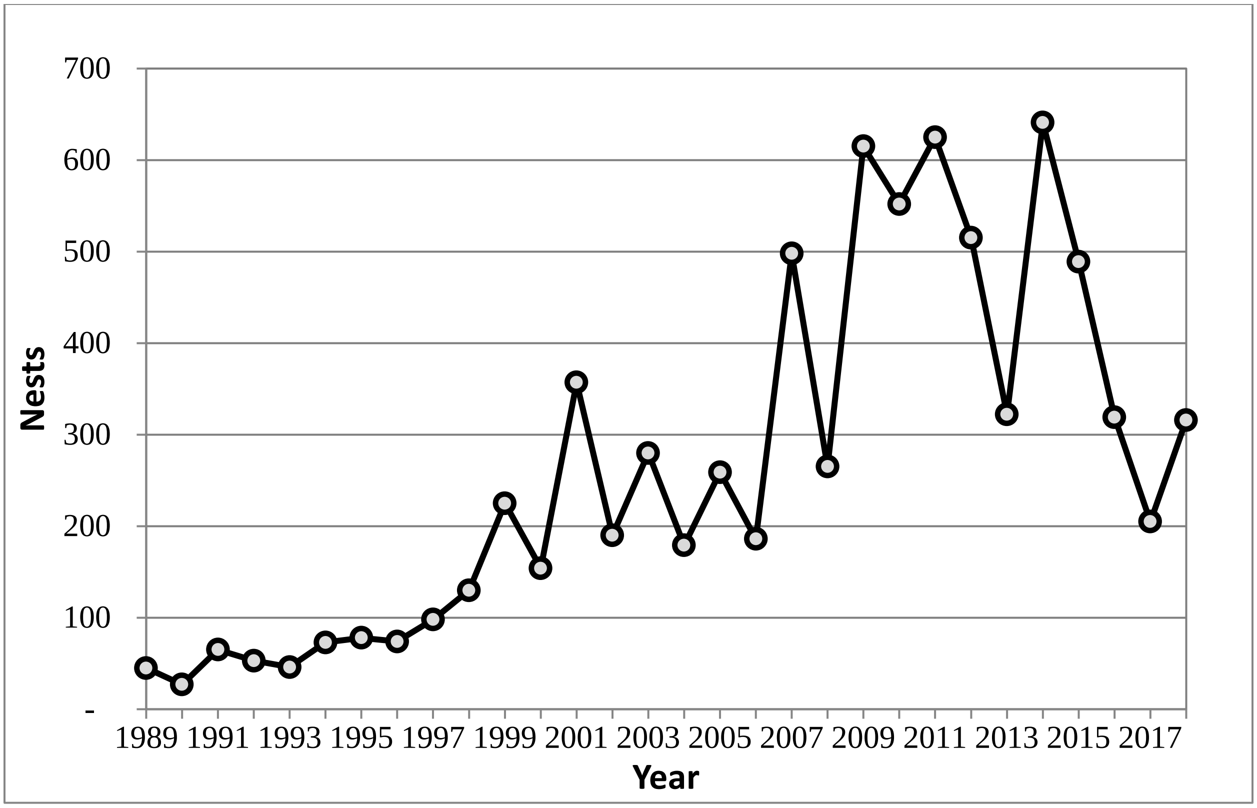

Figure 10. Leatherback Sea Turtle Nesting at Florida Index Beaches Since 1989

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

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

Because the available nesting information is inconsistent, it is difficult to estimate the total population size for Atlantic leatherbacks. Spotila et al. (1996) characterized the entire Western Atlantic population as stable at best and estimated a population of 18,800 nesting females. Spotila et al. (1996) further estimated that the adult female leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, was about 27,600 (considering both nesting and interesting females), with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the Turtle Expert Working Group (2007). The Turtle Expert Working Group (2007) also determined that at of 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. The latest review by NMFS and USFWS (2013b) suggests the leatherback nesting population is stable in most nesting regions of the Atlantic Ocean.

Threats
Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.3.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

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

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

As discussed in Section 3.3.1, 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 2007b). Several studies have shown leatherback distribution is influenced by jellyfish abundance (Houghton et al. 2006; Witt et al. 2007; Witt et al. 2006); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH oil spill on leatherback sea turtles are considered here. Available information indicates leatherback sea turtles (along with hawksbill turtles) were likely directly affected by the oil spill. Leatherbacks were documented in the spill area, but the number of affected leatherbacks was not estimated due to a lack of information compared to other species. But given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (Turtle Expert Working Group 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, it was concluded that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. Potential DWH-related impacts to leatherback sea turtles include direct oiling or contact with dispersants from surface and subsurface oil and dispersants, inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts likely occurred to leatherbacks, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event may be relatively low. Thus, a population-level impact may not have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

3.3.1.5 Hawksbill Sea Turtle

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

Species Description and Distribution
Hawksbill sea turtles are small- to medium-sized (99-150 lb on average [45-68 kg]) although females nesting in the Caribbean are known to weigh up to 176 lb (80 kg) (Pritchard et al. 1983). The carapace is usually serrated and has a tortoise-shell" coloring, ranging from dark to golden brown, with streaks of orange, red, and/or black. The plastron of a hawksbill turtle is typically yellow. The head is elongated and tapers to a point, with a beak-like mouth that gives the species its name. The shape of the mouth allows the hawksbill turtle to reach into holes and crevices of coral reefs to find sponges, their primary adult food source, and other invertebrates. The shells of hatchlings are 1.7 in (42 mm) long, are mostly brown, and are somewhat heart-shaped (Eckert 1995; Hillis and Mackay 1989; van Dam et al. 1990).

Hawksbill sea turtles have a circumtropical distribution and usually occur between latitudes 30°N and 30°S in the Atlantic, Pacific, and Indian Oceans. In the western Atlantic, hawksbills are widely distributed throughout the Caribbean Sea, off the coasts of Florida and Texas in the continental United States, in the Greater and Lesser Antilles, and along the mainland of Central America south to Brazil (Amos 1989; Groombridge and Luxmoore 1989; Lund 1985; Meylan and Donnelly 1999; NMFS and USFWS 1998; Plotkin and Amos 1990; Plotkin and Amos 1988). They are highly migratory and use a wide range of habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Adult hawksbill sea turtles are capable of migrating long distances between nesting beaches and foraging areas. For instance, a female hawksbill sea turtle tagged at Buck Island Reef National Monument (BIRNM) in St. Croix was later identified 1,160 mi (1,866 km) away in the Miskito Cays in Nicaragua (Spotila 2004).

Hawksbill sea turtles nest on sandy beaches throughout the tropics and subtropics. Nesting occurs in at least 70 countries, although much of it now only occurs at low densities compared to that of other sea turtle species (NMFS and USFWS 2007b). Meylan and Donnelly (1999) believe that the widely dispersed nesting areas and low nest densities is likely a result of overexploitation of previously large colonies that have since been depleted over time. The most significant nesting within the United States occurs in Puerto Rico and the U.S. Virgin Islands, specifically on Mona Island and BIRNM, respectively. Although nesting within the continental United States is typically rare, it can occur along the southeast coast of Florida and the Florida Keys. The largest hawksbill nesting population in the western Atlantic occurs in the Yucatán Peninsula of Mexico, where several thousand nests are recorded annually in the states of Campeche, Yucatán, and Quintana Roo (Garduño-Andrade et al. 1999; Spotila 2004). In the U.S. Pacific, hawksbills nest on main island beaches in Hawaii, primarily along the east coast of the island. Hawksbill nesting has also been documented in American Samoa and Guam. More information on nesting in other ocean basins may be found in the 5-year status review for the species (NMFS and USFWS 2007b).

Mitochondrial DNA studies show that reproductive populations are effectively isolated over ecological time scales (Bass et al. 1996). Substantial efforts have been made to determine the nesting population origins of hawksbill sea turtles assembled in foraging grounds, and genetic research has shown that hawksbills of multiple nesting origins commonly mix in foraging areas (ISTS 1995). Since hawksbill sea turtles nest primarily on the beaches where they were born, if a nesting population is decimated, it might not be replenished by sea turtles from other nesting rookeries (Bass et al. 1996).

Life History Information

Hawksbill sea turtles exhibit slow growth rates although they are known to vary within and among populations from a low of 0.4-1.2 in (1-3 cm) per year, measured in the Indo-Pacific (Chaloupka and Limpus 1997; Mortimer et al. 2003; Whiting 2000), to a high of 2 in (5 cm) or more per year, measured at some sites in the Caribbean (Diez and van Dam 2002; León and Diez 1999). Differences in growth rates are likely due to differences in diet and/or density of sea turtles at foraging sites and overall time spent foraging (Bjorndal and Bolten 2000; Chaloupka et al. 2004). Consistent with slow growth, age to maturity for the species is also long, taking between 20 and 40 years, depending on the region (Chaloupka and Musick 1997; Limpus and Miller 2000). Hawksbills in the western Atlantic are known to mature faster (i.e., 20 or more years) than sea turtles found in the Indo-Pacific (i.e., 30-40 years) (Boulon Jr. 1983; Boulon Jr. 1994; Diez and van Dam 2002; Limpus and Miller 2000). Males are typically mature when their length reaches 27 in (69 cm), while females are typically mature at 30 in (75 cm) (Eckert et al. 1992; Limpus 1992).

Female hawksbills return to the beaches where they were born (natal beaches) every 2-3 years to nest (van Dam et al. 1992; Witzell 1983) and generally lay 3-5 nests per season (Richardson et al. 1999). Compared with other sea turtles, the number of eggs per nest (clutch) for hawksbills can be quite high. The largest clutches recorded for any sea turtle belong to hawksbills (approximately 250 eggs per nest) (Hirth and Latif 1980), though nests in the U.S. Caribbean and Florida more typically contain approximately 140 eggs (USFWS hawksbill fact sheet, http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm). Eggs incubate for approximately 60 days before hatching (USFWS hawksbill fact sheet). Hatchling hawksbill sea turtles typically measure 1-2 in (2.5-5 cm) in length and weigh approximately 0.5 oz (15 g).

Hawksbills may undertake developmental migrations (migrations as immatures) and reproductive migrations that involve travel over many tens to thousands of miles (Meylan 1999a). Post-hatchlings (oceanic stage juveniles) are believed to live in the open ocean, taking shelter in floating algal mats and drift lines of flotsam and jetsam in the Atlantic and Pacific oceans (Musick and Limpus 1997) before returning to more coastal foraging grounds. In the Caribbean, hawksbills are known to almost exclusively feed on sponges (Meylan 1988; van Dam and Díez 1997), although at times they have been seen foraging on other food items, notably corallimorphs and zooanthids (León and Díez 2000; Mayor et al. 1998; van Dam and Díez 1997).

Reproductive females undertake periodic (usually non-annual) migrations to their natal beaches to nest and exhibit a high degree of fidelity to their nest sites. Movements of reproductive males are less certain, but are presumed to involve migrations to nesting beaches or to courtship stations along the migratory corridor. Hawksbills show a high fidelity to their foraging areas as well (van Dam and Diez 1998). Foraging sites are typically areas associated with coral reefs, although hawksbills are also found around rocky outcrops and high energy shoals which are optimum sites for sponge growth. They can also inhabit seagrass pastures in mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent (Bjorndal 1997; van Dam and Diez 1998).

Status and Population Dynamics

There are currently no reliable estimates of population abundance and trends for non-nesting hawksbills at the time of this consultation; therefore, nesting beach data is currently the primary information source for evaluating trends in global abundance. Most hawksbill populations around the globe are either declining, depleted, and/or remnants of larger aggregations (NMFS and USFWS 2007b). The largest nesting population of hawksbills occurs in Australia where approximately 2,000 hawksbills nest off the northwest coast and about 6,000-8,000 nest off the Great Barrier Reef each year (Spotila 2004). Additionally, about 2,000 hawksbills nest each year in Indonesia and 1,000 nest in the Republic of Seychelles (Spotila 2004). In the United States, hawksbills typically laid about 500-1,000 nests on Mona Island, Puerto Rico in the past (Diez and van Dam 2007), but the numbers appear to be increasing, as the Puerto Rico Department of Natural and Environmental Resources counted nearly 1,600 nests in 2010 (PRDNER nesting data). Another 56-150 nests are typically laid on Buck Island off St. Croix (Meylan 1999b; Mortimer and Donnelly 2008). Nesting also occurs to a lesser extent on beaches on Culebra Island and Vieques Island in Puerto Rico, the mainland of Puerto Rico, and additional beaches on St. Croix, St. John, and St. Thomas, U.S. Virgin Islands.

Mortimer and Donnelly (2008) reviewed nesting data for 83 nesting concentrations organized among 10 different ocean regions (i.e., Insular Caribbean, Western Caribbean Mainland, Southwestern Atlantic Ocean, Eastern Atlantic Ocean, Southwestern Indian Ocean, Northwestern Indian Ocean, Central Indian Ocean, Eastern Indian Ocean, Western Pacific Ocean, Central Pacific Ocean, and Eastern Pacific Ocean). They determined historic trends (i.e., 20-100 years ago) for 58 of the 83 sites, and also determined recent abundance trends (i.e., within the past 20 years) for 42 of the 83 sites. Among the 58 sites where historic trends could be determined, all showed a declining trend during the long-term period. Among the 42 sites where recent (past 20 years) trend data were available, 10 appeared to be increasing, 3 appeared to be stable, and 29 appeared to be decreasing. With respect to regional trends, nesting populations in the Atlantic (especially in the Insular Caribbean and Western Caribbean Mainland) are generally doing better than those in the Indo-Pacific regions. For instance, 9 of the 10 sites that showed recent increases are located in the Caribbean. Buck Island and St. Croix’s East End beaches support 2 remnant populations of between 17-30 nesting females per season (Hillis and Mackay 1989; Mackay 2006). While the proportion of hawksbills nesting on Buck Island represents a small proportion of the total hawksbill nesting occurring in the greater Caribbean region, Mortimer and Donnelly (2008) report an increasing trend in nesting at that site based on data collected from 2001-2006. The conservation measures implemented when BIRNM was expanded in 2001 most likely explains this increase.

Nesting concentrations in the Pacific Ocean appear to be performing the worst of all regions despite the fact that the region currently supports more nesting hawksbills than either the Atlantic or Indian Oceans (Mortimer and Donnelly 2008). While still critically low in numbers, sightings of hawksbills in the eastern Pacific appear to have been increasing since 2007, though some of that increase may be attributable to better observations (Gaos et al. 2010). More information about site-specific trends can be found in the most recent 5-year status review for the species (NMFS and USFWS 2007b).

Threats

Hawksbills are currently subjected to the same suite of threats on both nesting beaches and in the marine environment that affect other sea turtles (e.g., interaction with federal and state fisheries, coastal construction, oil spills, climate change affecting sex ratios) as discussed in Section 3.3.1. There are also specific threats that are of special emphasis, or are unique, for hawksbill sea turtles discussed in further detail below.

While oil spill impacts are discussed generally for all species in Section 3.3.1, specific impacts of the DWH spill on hawksbill turtles have been estimated. Hawksbills made up 2.2% (8,850) of small juvenile sea turtle (of those that could be identified to species) exposures to oil in offshore areas, with an estimate of 615 to 3,090 individuals dying as a result of the direct exposure (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). No quantification of large benthic juveniles or adults was made. Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts occurred to hawksbills, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event is relatively low, and thus a population-level impact is not believed to have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

The historical decline of the species is primarily attributed to centuries of exploitation for the beautifully patterned shell, which made it a highly attractive species to target (Parsons 1972). The fact that reproductive females exhibit a high fidelity for nest sites and the tendency of hawksbills to nest at regular intervals within a season made them an easy target for capture on nesting beaches. The shells from hundreds of thousands of sea turtles in the western Caribbean region were imported into the United Kingdom and France during the nineteenth and early twentieth centuries (Parsons 1972). Additionally, hundreds of thousands of sea turtles contributed to the region's trade with Japan prior to 1993 when a zero quota was imposed (Milliken and Tokunaga 1987), as cited in Bräutigam and Eckert (2006).

The continuing demand for the hawksbills' shells as well as other products derived from the species (e.g., leather, oil, perfume, and cosmetics) represents an ongoing threat to its recovery. The British Virgin Islands, Cayman Islands, Cuba, Haiti, and the Turks and Caicos Islands (United Kingdom) all permit some form of legal take of hawksbill sea turtles. In the northern Caribbean, hawksbills continue to be harvested for their shells, which are often carved into hair clips, combs, jewelry, and other trinkets (Márquez M. 1990; Stapleton and Stapleton 2006). Additionally, hawksbills are harvested for their eggs and meat, while whole, stuffed sea turtles are sold as curios in the tourist trade. Hawksbill sea turtle products are openly available in the Dominican Republic and Jamaica, despite a prohibition on harvesting hawksbills and their eggs (Fleming 2001). Up to 500 hawksbills per year from 2 harvest sites within Cuba were legally captured each year until 2008 when the Cuban government placed a voluntary moratorium on the sea-turtle fishery (Carillo et al. 1999; Mortimer and Donnelly 2008). While current nesting trends are unknown, the number of nesting females is suspected to be declining in some areas (Carillo et al. 1999; Moncada et al. 1999). International trade in the shell of this species is prohibited between countries that have signed the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), but illegal trade still occurs and remains an ongoing threat to hawksbill survival and recovery throughout its range.

Due to their preference to feed on sponges associated with coral reefs, hawksbill sea turtles are particularly sensitive to losses of coral reef communities. Coral reefs are vulnerable to destruction and degradation caused by human activities (e.g., nutrient pollution, sedimentation, contaminant spills, vessel groundings and anchoring, recreational uses) and are also highly sensitive to the effects of climate change (e.g., higher incidences of disease and coral bleaching) (Crabbe 2008; Wilkinson 2004). Because continued loss of coral reef communities (especially in the greater Caribbean region) is expected to impact hawksbill foraging, it represents a major threat to the recovery of the species.

4. ENVIRONMENTAL BASELINE

This section is a description of the effects of past and ongoing human and natural factors leading to the current status of the species, their habitat (including designated critical habitat), and ecosystem, within the action area. The environmental baseline does not include the effects of the action under review in this Opinion.

By regulation, environmental baselines for biological opinions include the past and present impacts of all state, federal, or private actions and other human activities in the action area. We identify the anticipated impacts of all proposed federal projects in the specific action area of the consultation at issue, that have already undergone formal or early Section 7 consultation as well as the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02).

Focusing on the impacts of the activities in the action area specifically, allows us to assess the prior experience and state (or condition) of the endangered and threatened individuals, and areas of designated critical habitat that occur in an action area, and that will be exposed to effects from the action under consultation. This is important because, in some phenotypic states or life history stages, 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. The same is true for localized populations of endangered and threatened species: the consequences of changes in the fitness or performance of individuals on a population's status depends on the prior state of the population.

The following subsections are synopses of the actions and the effects these actions have had or are having on loggerhead sea turtles (NWA DPS), Kemp's ridley sea turtles, green sea turtles (NA and SA DPSs), hawksbill sea turtles, and leatherback sea turtles within the action area.

4.1. Federal Actions

4.1.1 ESA Section 10 Permits

Sea turtles are the focus of research activities authorized by Section 10 permits under the ESA. The ESA allows the issuance of permits to take listed species for the purposes of scientific research and enhancement (Section 10(a)(1)(A)). In addition, the ESA allows for NMFS to enter into cooperative agreements with states, developed under Section 6 of the ESA, to assist in recovery actions of listed species. Prior to issuance of these authorizations, the proposal must be reviewed for compliance with Section 7 of the ESA. Per a search of the National Oceanic and Atmospheric Administration (NOAA) Fisheries Authorizations and Permits for Protected Species (APPS) database by the consulting biologist on June 17, 2019, there were 10 active Section 10(a)(1)(A) scientific research permits applicable to loggerhead, green, Kemp's ridley, leatherback, and hawksbill sea turtles within the action area.[8] These permits allow the capture, handling, sampling, and release of these turtle species (all life stages except hatchlings) and range in purpose from reducing bycatch in commercial fisheries to gaining better scientific knowledge.

4.1.2 Vessel Activity and Operations

Potential sources of adverse effects from federal vessel activity and operations in the action area include operations of the USCG and the United States Navy (USN). Through the Section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species. Refer to the Biological Opinions for the USCG (NMFS 1995; NMFS 1996) and the USN (NMFS 1996;

[8] https://apps.nmfs.noaa.gov/index.cfm

NMFS 1997a; NMFS 2013) for details on the scope of vessel operations for these agencies and conservation measures implemented as standard operating procedures.

4.1.3 Dredging

The construction and maintenance of federal navigation channels and sand mining sites ("borrow areas") conducted by the USACE has been identified as a source of sea turtle mortality. Hopper dredges have been known to entrain and kill sea turtles as a result of the suction dragheads of the advancing dredge. Entrainment events most likely occur when hopper dredge dragheads approach an animal that is oriented on the bottom and either resting or foraging and moving at minimal speed. In most cases, the entrainment scenario occurs when the operating environment presents challenges for the turtle deflector to operate as designed and the operator is not able to keep the draghead(s) fixed on the bottom. Similarly, entrainment can occur when a sea turtle burrows into the substrate or is within a hole/trench/depression that the draghead moves over. Entrained sea turtles rarely survive.

NMFS completed a regional Biological Opinion on the impacts of USACE's South Atlantic coast hopper-dredging operations in 1997 for dredging in the USACE's South Atlantic Division (NMFS 1997b). The regional Biological Opinion on South Atlantic hopper dredging (SARBO) of navigational channels and borrow areas determined that hopper dredging would not adversely affect leatherback sea turtles in the South Atlantic Division (i.e., coastal states of North Carolina through Key West, Florida). The Opinion determined hopper dredging in the South Atlantic Division would adversely affect 4 sea turtle species (i.e., green, hawksbill, Kemp's ridley, and loggerhead), but it would not jeopardize their continued existence. Reinitiation of consultation on the 1997 SARBO has been triggered for a number of reasons, including listing of new species and designation of critical habitat that may be affected by these dredging activities.

NMFS completed a Biological Opinion (NMFS 2018) on the impacts of a long-term beach and inlet management plan in North Carolina by the USACE and the Bureau of Ocean Energy Management (BOEM). The Opinion determined that the dredging activities are not likely to jeopardize the continued existence of green sea turtle (NA or SA DPS) and loggerhead sea turtle (NWA DPS).

There are no other dredging Opinions that address activities within the action area as per a review of the NMFS PRD's completed consultation database and a search of NMFS's records by the consulting biologist on June 17, 2019.

4.1.4 Beach Nourishment

The USACE issues Clean Water Act permits for disposal of material in navigable waters of the United States, including beach nourishment. The activity of beach nourishment, particularly when impacts include the loss of nearshore hard bottom habitat along the east coast of Florida, has been documented to result in injury and death of juvenile green sea turtles. Juvenile green turtles are known to utilize these high-energy, dynamic habitats for foraging and as refuge, and show a preference for this habitat even when abundant deeper-water sites are available. The loss of such limited habitat, especially when considering the cumulative loss as a result of beach nourishment activities occurring along the entire range of the habitat and continually over time, is expected to result in loss of foraging opportunities and protective refuge. The stresses are also expected to contribute to mortality of individuals already in poor condition as a result of disease or other factors. Beach nourishment permitted by the USACE also often involves use of a hopper dredge to collect nourishment material, thus posing another route of adverse effects to sea turtles.

NMFS completed a Biological Opinion (SER-2017-18882 Bogue Banks Master Beach Nourishment Plan) on the impacts of a long-term beach and inlet management plan in North Carolina by the USACE and the Bureau of Ocean Energy Management (BOEM). The Opinion determined that the beach nourishment activities are not likely to jeopardize the continued existence of green sea turtle (NA or SA DPS) and loggerhead sea turtle (NWA DPS).

There are no beach nourishment Opinions that address activities within the action area as per a review of the NMFS PRD's completed consultation database and a search of NMFS's records by the consulting biologist on June 17, 2019.

4.1.5 Fisheries Monitoring

NMFS Integrated Fisheries Independent Monitoring Activities in the Southeast (Atlantic) Region promotes and funds projects conducted by the SEFSC and other NMFS partners to collect fisheries independent data. The various projects use a variety of gear (e.g., trawls, nets, etc.) to conduct fishery research. NMFS issued an Opinion on the continued authorization and implementation of these projects on May 9, 2016 (SER-2009-07541). The Opinion determined the continued authorization and implementation of these projects would adversely affect ESA-listed sea turtle species, but it would not jeopardize their continued existence.

4.1.6 Federally Managed Fisheries

Sea turtles are adversely affected by fishing gears and vessels used throughout the continental shelf of the action area. Hook-and-line gear, trawl, and pot fisheries have all been documented as interacting with sea turtles. Formal Section 7 consultations have been conducted on the fisheries discussed in the following sections, occurring at least in part within the action area; these fisheries use gear known to adversely affect listed sea turtle species. A brief summary of each fishery is provided below, but more detailed information can be found in the respective Opinions.

4.1.6.1 <u>Finfish Fisheries</u>

In February 2012, NMFS issued an Opinion on the continued authorization of the Atlantic dolphin-wahoo fishery (SER-2012-00410). The Opinion concluded the fishery may adversely affect, but would not jeopardize the continued existence of any ESA-listed sea turtle species. On June 11, 2018, NMFS reinitiated consultation to address the listing of new species, including the North and South Atlantic green sea turtle DPSs, and concluded that allowing this fishery to continue during the reinitiating period will not jeopardize the continued existence of these DPSs.

In 2012, NMFS issued an Opinion on the continued authorization of Highly Migratory Species Atlantic shark fisheries (NMFS 2012a). This commercial fishery uses bottom longline and gillnet gear. The recreational sector of the fishery uses only hook-and-line gear. To protect declining shark stocks, the proposed action seeks to greatly reduce the fishing effort in the commercial component of the fishery. These reductions are likely to greatly reduce the interactions between the commercial component of the fishery and sea turtles. The Opinion concluded that ESA-listed sea turtle species may be adversely affected by operation of the fishery but that the proposed action was not expected to jeopardize the continued existence of any of these species. NMFS reinitiated consultation to address the listing of new species, including the North and South Atlantic green sea turtle DPSs, and concluded that allowing this fishery to continue during the reinitiating period will not jeopardize the continued existence of these DPSs.

The Atlantic bluefish fishery has been operating in the U.S. Atlantic for at least the last half century, although its popularity did not heighten until the late 1970s and early 1980s (MAFMC and ASMFC 1998). The gears used include otter trawls, gillnets, and hook-and-line. The majority of commercial fishing activity in the north Atlantic and mid-Atlantic occurs in the late spring to early fall, when bluefish are most abundant in these areas (40th Northeast Regional Stock Assessment Workshop (40th SAW) 2005). Formal consultations on the fishery have been conducted in 1999, 2010, and most recently in December 2013. The 2013 consultation included an evaluation of the effects of the fishery on ESA-listed sea turtles. The bluefish fishery was considered as part of a larger “batched” consultation which evaluated the effects of the (1) Northeast multispecies, (2) monkfish, (3) spiny dogfish, (4) Atlantic bluefish, (5) Northeast skate complex, (6) Atlantic mackerel/squid/butterfish, and (7) summer flounder/scup/BSB fisheries. The consultation concluded that the continued operation of the Atlantic bluefish fishery was likely to adversely affect, but not jeopardize, the continued existence of any ESA-listed sea turtle species. NMFS reinitiated consultation to address the listing of new species, including the North and South Atlantic green sea turtle DPSs, and concluded that allowing this fishery to continue during the reinitiating period will not jeopardize the continued existence of these DPSs.

NMFS completed a Section 7 consultation on the continued authorization of the coastal migratory pelagic resources fishery in the Gulf of Mexico and South Atlantic in 2015 (NMFS 2015). In the Gulf of Mexico and South Atlantic, commercial fishers target king and Spanish mackerel with hook-and-line (i.e., handline, rod-and-reel, and bandit), gillnet, and cast net gears. Recreational fishers in both areas use only rod-and-reel. Trolling is the most common hook-and-line fishing technique used by both commercial and recreational fishers. Although run-around gillnets accounted for the majority of the king mackerel catch from the late 1950s through 1982, in 1986, and in 1993, handline gear has been the predominant gear used in the commercial king mackerel fishery since 1993 (NMFS 2015). The consultation concluded that the continued operation of the coastal migratory pelagic resources fishery in the Gulf of Mexico and South Atlantic was likely to adversely affect, but not jeopardize, the continued existence of any ESA-listed sea turtles species. NMFS reinitiated consultation because of the North Atlantic and South Atlantic DPS of green sea turtle listing and completed an amendment to the biological opinion on November 18, 2017 that concluded the authorization of this fishery is not likely to jeopardize the continued existence of either DPS of green sea turtle.

NOAA Fisheries Service completed an Opinion on the South Atlantic snapper-grouper fishery entitled: “The Continued Authorization of Snapper-Grouper Fishing in the U.S. South Atlantic Exclusive Economic Zone (EEZ) as Managed Under the Snapper-Grouper Fishery Management Plan of the South Atlantic Region (SGFMP), including Amendment 16 to the SGFMP” (SER-

2016-17768) in 2016. The Opinion concluded the continued authorization of the fishery is not likely to jeopardize the continued existence of any ESA-listed sea turtles.

4.1.6.2 Southeastern Shrimp Trawl Fisheries

Southeastern U.S. shrimp fisheries target primarily brown, white, and pink shrimp in inland waters and estuaries through the state-regulated territorial seas and in federal waters of the EEZ. As sea turtles rest, forage, or swim on or near the bottom, these species are captured by shrimp trawls that are pulled along the bottom. In 1990, the National Research Council concluded that the southeastern U.S. shrimp trawl fisheries affected more sea turtles than all other activities combined and was the most significant anthropogenic source of sea turtle mortality in the U.S. waters, in part due to the high reproductive value of turtles taken in this fishery (National Research Council 1990b).

On May 9, 2012, NMFS completed an Opinion that analyzed the continued implementation of the sea turtle conservation regulations and the continued authorization of the Southeast shrimp fisheries in federal waters under the Magnuson-Stevens Act (MSA) (NMFS 2012a). The Opinion also considered a proposed amendment to the sea turtle conservation regulations that would withdraw the alternative tow time restriction at 50 CFR 223.206(d)(2)(ii)(A)(*3*) for skimmer trawls, pusher-head trawls, and wing nets (butterfly trawls) and instead require all of these vessels to use TEDs. The Opinion concluded that the proposed action would not jeopardize the continued existence of any sea turtle species. The Opinion requires NMFS to minimize the impacts of incidental takes through monitoring of shrimp effort and regulatory compliance levels, conducting TED training and outreach, and continuing to research the effects of shrimp trawling on listed species. Subsequent to the completion of this opinion, NMFS withdrew the proposed amendment to require TEDs in skimmer trawls, pusher-head trawls, and wing nets. Consequently, NMFS reinitiated consultation on November 26, 2012. Consultation was completed in April 2014 and determined the continued implementation of the sea turtle conservation regulations and the continued authorization of the southeastern U.S. shrimp fisheries in federal waters under the MSA was not likely jeopardize the continued existence of any ESA-listed sea turtle species. NMFS has reinitiated consultation to address the listing of new species, including the North and South Atlantic green sea turtle DPSs, and concluded that allowing these fisheries to continue during the reinitiating period will not jeopardize the continued existence of these DPSs.

4.1.7 Existing Artificial Reef Sites

These USACE permitted artificial reefs are the subject of this consultation and have affected the environmental baseline for sea turtles in the action area. There are currently 43 existing ocean reefs and 25 existing estuarine reefs maintained by the state of North Carolina by the North Carolina Division of Marine Fisheries (NCDMF). Current materials and the approximate amount accumulated over all existing North Carolina reefs as of 2016 (DMF Reef Guide) are summarized in Table 6. More than approximately 50% of the existing reef sites contain “high-relief” material.

Table 6. Summary of Material Types and Occurrence at Current Reef Sites

Material Type	Approximate Number of Reef Sites Material is Present
Train boxcar	13
Reef Ball	32
Piling	3
Concrete – pipes, crushed, consolidated, recycled, rubble	37
Vessel/tug/oiler/dredge/barge	24
Aircraft	5
Bridge Rubble/frames/railing	7
Concrete Boxes	2
High Profile Units	1
Manhole Sections and Risers	10
Buoy Anchors	1
Fiberglass Domes and Boat Molds	4
Tires	4
Metal Containers	1
Atlantic Pods	4
Waffle-Crete Units	2
Floating Drydock	1
Boiler Pieces	1
Limestone Marl/Riprap	13
Rock/oyster shells/clams	4
"H" Units	2

4.2. State or Private Actions

4.2.1 Maritime Industry

Private and commercial vessels operating in the action area have the potential to interact with ESA-listed species. The effects of vessels on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. Commercial traffic and recreational pursuits can also adversely affect sea turtles through propeller and boat strikes. The Sea Turtle Stranding and Salvage Network (STSSN) include many records of vessel interaction with sea turtles where there are high levels of vessel traffic. The extent of the problem is difficult to assess because we cannot know whether the majority of sea turtles are struck pre- or post-mortem. It is important to note that minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so it is more likely to become vulnerable to effects such as entanglements or predation. NMFS and the USCG have completed several formal consultations on individual marine events that may affect sea turtles.

4.2.2 Coastal Development

Beachfront development, lighting, and beach erosion control all are ongoing activities along the North Carolina coastline. These activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea. Nighttime human activities along nesting beaches may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. However, more and more coastal counties are adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting.

4.2.3 State Fisheries

Recreational fishing as regulated by the State of North Carolina can affect protected species or their habitats within the action area. Pressure from recreational fishing in and adjacent to the action area is likely to continue. Observations of state recreational fisheries have shown that loggerhead sea turtles are known to bite baited hooks and frequently ingest the hooks. Hooked sea 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 (NMFS 2001). 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 Kemp's ridley and loggerhead sea turtles can be found in the TEWG reports (Turtle Expert Working Group 1998; Turtle Expert Working Group 2000).

NMFS completed a Biological Opinion (SER-2017-18675 Yaupon Fishing Pier) on the impacts of the restoration and extension of a public fishing pier in North Carolina by the USACE and the U.S. Department of Homeland Security Federal Emergency Management Agency (FEMA). The Opinion determined that the restoration and extension of the public fishing pier is not likely to jeopardize the continued existence of the NA or SA DPSs of green sea turtle, Kemp's ridley sea turtle, or the NWA DPS of loggerhead sea turtle.

There have been no fishing pier biological opinions addressing activities within the action area as per a review of the NMFS PRD's completed consultation database and a search of PCTS and ECO records by the consulting biologist on June 17, 2019.

In August of 2007, NMFS issued a regulation (72 FR 43176, August 3, 2007) to require any fishing vessels subject to the jurisdiction of the United States to take observers upon NMFS's request. The purpose of this measure is to learn more about sea turtle interactions with fishing operations, to evaluate existing measures to reduce sea turtle takes, and to determine whether additional measures to address prohibited sea turtle takes may be necessary.

4.3. Climate Change

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

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

4.4. Marine Pollution and Environmental Contamination

Sources of pollutants along the coastal areas include atmospheric loading of Polychlorinated Biphenyls (PCBs), stormwater runoff from coastal towns and cities into rivers and canals emptying into bays and the ocean, and groundwater and other discharges (Vargo et al. 1986). Nutrient loading from land-based sources such as coastal community discharges is known to stimulate plankton blooms in closed or semi-closed estuarine systems (Bowen and Valiela 2001; Rabalais et al. 2002). The effects on larger embayments are unknown. Although pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo et al. 1986), the impacts of many other anthropogenic toxins have not been investigated.

Coastal runoff, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, increased under water noise and boat traffic can degrade marine habitats used by sea turtles (Colburn et al. 1996). The development of marinas and docks in inshore waters can negatively impact nearshore habitats. An increase in the number of docks built increases boat and vessel traffic. Fueling facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive estuarine and coastal habitats. Although these contaminant concentrations do not likely affect the more pelagic waters, the species analyzed in this Opinion travel between near shore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles.

There are studies on organic contaminants and trace metal accumulation in green and leatherback sea turtles (Aguirre et al. 1994; Caurant et al. 1999; Corsolini et al. 2000). Mckenzie et al. (1999) measured concentrations of PFCs and organochlorine pesticides (such as Dichlorodiphenyltrichloroethane [DDT]) in sea turtle tissues collected from the Mediterranean (Cyprus, Greece) and European Atlantic waters (Scotland) between 1994 and 1996. Omnivorous loggerhead turtles had the highest organochlorine contaminant concentrations in all the tissues sampled, including those from green and leatherback turtles (Storelli et al. 2008). Dietary preferences were likely the main differentiating factor among species. Decreasing lipid contaminant burdens with turtle size were observed in green turtles, most likely attributable to a change in diet with age. 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).

4.5. Stochastic Events

Stochastic (i.e., random) events, such as hurricanes, occur in North Carolina and can affect the action area. These events are by nature unpredictable, and their effect on the recovery of ESA-listed sea turtles 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 cold snap, can injure or kill these species.

4.6. Conservation and Recovery Actions Benefiting Sea Turtles

NMFS has implemented a number of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for Atlantic Highly Migratory Species (HMS) and Gulf of Mexico reef fish fisheries, and Turtle Excluder Device (TED) requirements for the southeastern shrimp fisheries. TEDs and other bycatch reduction device requirements may reduce sea turtle bycatch in Southeast trawl fisheries (Atlantic Sturgeon Status Review Team 2007). NMFS has required the use of TEDs in southeast United States shrimp trawls since 1989 and in summer flounder trawls in the mid-Atlantic area (south of Cape Charles, Virginia) since 1992 to reduce the potential for incidental mortality of sea turtles in commercial trawl fisheries. These regulations have been refined over the years to ensure that TED effectiveness is maximized through more widespread use, and proper placement, installation, floatation, and configuration (e.g., width of bar spacing). NMFS has also been working to develop a TED, which can be effectively used in a type of trawl known as a flynet, which is sometimes used in the mid-Atlantic and Northeast fisheries to target sciaenids and bluefish. A top-opening flynet TED was certified in the summer of 2007, but experiments are still ongoing to certify a bottom-opening TED. All of these changes may lead to greater conservation benefits for ESA-listed sea turtle species.

5. EFFECTS OF THE ACTION ON LISTED SPECIES

In this section of our Opinion, we assess the direct and indirect effects of the proposed 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 7.0. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Data are limited, so we are often forced to make assumptions to overcome the limits in our knowledge. Sometimes, the best available information may include a range of values for a particular aspect under consideration, or different analytical approaches may be applied to the same data set. In those cases, the uncertainty is resolved in favor of the species (House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)). NMFS generally selects the value 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.

As discussed in Section 3, sea turtles are the only ESA-listed species that may be adversely affected by the proposed action. In this section we discuss the components of the action that may adversely affect sea turtles.

5.1. Potential Effects to ESA-listed Sea Turtles from Entanglement

NMFS believes that the presence of high-relief artificial reef material is likely to adversely affect loggerhead sea turtle (NWA DPS), green sea turtle (NA and SA DPSs), Kemp's ridley sea turtle, leatherback sea turtle, and hawksbill sea turtle.

Approach to Assessment

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

Since the project proposes to deploy high-relief, complex artificial reef material such as vessels, we anticipate adverse effects on sea turtles from entanglement and drowning in monofilament and other entangling gear that accumulates on the reef material.

There is adequate information that high-relief and complex materials such as vessels, aircraft, bridge spans, decommissioned oil rigs, and other metal structures, etc., may present significant entanglement risk to sea turtles over time. This is likely due to the ease of locating vessels deployed as artificial reefs by even inexperienced fishers (versus smaller, low-relief sites), the relatively large footprint that allow fishers to remain over the site and fish, the complicated vertical and horizontal surfaces that facilitate the loss of fishing gear, as well as other potential factors.

In general, due to the absence of monofilament immediately following an artificial reef deployment, we expect the risk of entanglement to be extremely low for some period of years. However, as time passes and monofilament line accumulates, the probability of an entanglement event increases. Also, the longer the accumulated line is present, the greater the chance that a sea turtle will encounter it. The rate of monofilament accumulation and the time it takes to reach the level where we might anticipate an entanglement-related mortality likely varies significantly due to the factors previously mentioned. As time passes, the integrity of the vessel 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. Figure 11 below illustrates the theoretical lifespan of a vessel deployed as an artificial reef over 100 years. While the curve of entanglement probability will vary by site for various reasons, the point is that entanglement risk is not static.

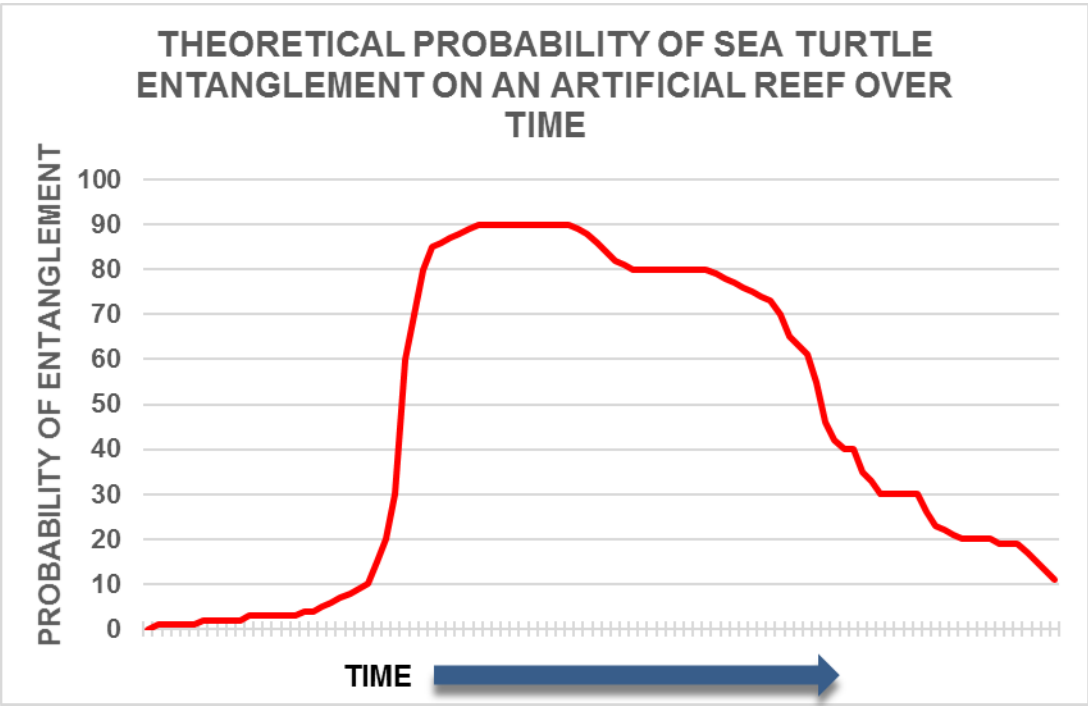

Figure 11. Theoretical Probability of Sea Turtle Entanglement on an Artificial Reef Over Time

Based on available information presented in (Barnette 2017) and sea turtle stranding data, we anticipate adult loggerhead sea turtles will be the sea turtle species primarily associated with entanglement events on artificial reef material as a result of the proposed action. This is likely due to the loggerhead sea turtle's 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 internesting 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 internesting interval was spent at depths less than 66 ft. While loggerheads have been documented diving to depths exceeding 760 ft (Sakamoto et al. 1990), other studies have demonstrated the majority of dives are occurring at much shallower depths. For instance, Arendt et al. (2012) documented most dives were conducted shallower than 160 ft, and were typically between 65-130 ft, when looking at male loggerhead sea turtles off the southeastern U.S.

However, one of the authors of this study noted that one of the limitations about diving behavior is that a lot of the depths reflect where animals were captured and individual animal preferences, and do not reflect comprehensive diving behavior across the species as a whole (M. Arendt, SCDNR, pers. comm. with NMFS Biologist M. Barnette). While it may be useful to discount a depth below which we believe the threat of entanglement from monofilament on artificial reefs is unlikely to occur based on sea turtle diving behavior, the available literature is insufficient to support an adequately informed decision regarding the appropriate depth threshold. Furthermore, the majority of waters in the southeastern U.S. where artificial reefs may be deployed would likely be much shallower than this theoretical depth delineation and, therefore, not be exempt from consideration of these effects. As a result, all complex, high-relief materials deployed as an artificial reef, regardless of depth, are included in our analysis.

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

Barnette (2017) documents that sites submerged for more than 120 years may still accumulate monofilament and result in sea turtle mortalities due to entanglement events. Given the remaining structure on that site (Barnette 2017), it is likely to persist for another 30 years. In contrast to other complex high-relief artificial reef material, aircraft generally deteriorate much faster due to differences in construction and materials. Barnette (2017) included an example of an entanglement event from an aircraft that sank 65 years earlier. Only trace amounts were left of that aircraft, and we suspect it won't be exposed much longer. Therefore, for purposes of this analysis, we will use an effective lifespan of 150 years for vessels, decommissioned oil rigs, bridge spans, etc., and 85 years for aircraft and smaller complex and/or high-relief materials.

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 at the same time. The author also noted that evidence of sea turtle entanglement events is ephemeral, and the absence of evidence of entanglement should not be viewed as evidence that entanglements have not occurred. Perhaps some complex, high-relief artificial reefs will never result in a sea turtle mortality due to entanglement, but given the available information, we take a risk-averse approach and consider all vessels, metal structures, and aircraft 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. Given the available information, our informed judgement, and taking a risk-averse approach, we will assume 1 sea turtle entanglement event per year on a "mature" artificial reef site (i.e., a site that has accumulated sufficient line to present a lethal threat). Therefore, after 25 years, we conservatively assume any high-relief artificial structure will result in 1 sea turtle mortality per year due to entanglement. Serious entanglement will effectively anchor a sea turtle to the artificial reef and prevent it from reaching the surface to breath, resulting in sea turtle mortality due to drowning (i.e., forced submergence). Numerous entanglement examples are documented in Barnette (2017). We consider this effect to be ongoing for the next 75 years for vessels, decommissioned oil rigs, bridge spans, and similar materials. After that point, we anticipate entanglement risk will be reduced on average due to material deterioration, subsidence, etc., and that entanglement risk over the next 50 years of the material's effective lifespan result in a sea turtle mortality every 3 years. Table 7 shows the 25 new vessels proposed over the next 7 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, etc.[9]

Table 7. Future NCDMF Reef Projects using High-Relief Material

Year	Anticipated Reef Sites	Material
2019	AR-165, AR-250, AR-255, AR-368	4 vessels
2020	AR-368, AR-430	2 vessels
2021	AR-430, AR-305	4 vessels
2022	AR-305	3 vessels
2023	Unknown	3 vessels
2024	Unknown	3 vessels
2025	Unknown	3 vessels
2026	Unknown	3 vessels

If 1 vessel results in 0 sea turtle mortalities for the first 25 years, transitions to 1 sea turtle mortality each year for the next 75 years, and changes to 1 sea turtle mortality every 3 years for the following 50 years, we would calculate the overall sea turtle take for the 150-year period as a result of the projected deployments for this project by year as:

Year 2019 (4 vessels, 4 reef sites): 92 sea turtle mortalities*4 vessel deployments in 1 year = **368 sea turtle takes**

Year 2020 (2 vessels, 2 reef sites): 92 sea turtle mortalities*2 vessel deployments in 1 year = **184 sea turtle takes**

Year 2021 (4 vessels, 2 reef sites): 92 sea turtle mortalities*4 vessel and complex material deployments in 1 year = **368 sea turtle takes**

Year 2022 (3 vessels, 1 reef site): 92 sea turtle mortalities*3 vessel and complex material deployments in 1 year = **276 sea turtle takes**

[9] In comparison, anticipated take estimate for an aircraft and smaller complex materials would follow a similar, but truncated course for the reasons discussed above. We estimate that after a 25-year period for monofilament to accumulate to sufficient levels, sea turtles may become entangled every 3 years for the next 60 years. This would result in an estimated take of 20 sea turtles over 85 years.

Year 2023 (3 vessels, reef site[s]): 92 sea turtle mortalities*3 vessel deployments in 1 year = **276 sea turtle takes**

Year 2024 (3 vessels, unknown reef site[s]): 92 sea turtle mortalities*3 vessel deployments in 1 year = **276 sea turtle takes**

Year 2025 (3 vessels, unknown reef site[s]): 92 sea turtle mortalities*3 vessel deployments in 1 year = **276 sea turtle takes**

Year 2026 (3 vessels, unknown reef site[s]): 92 sea turtle mortalities*3 vessel deployments in 1 year = **276 sea turtle takes**

In total, the number of sea turtle takes over the course of 150 years as a result of the deployed vessels over the next 7 years would equal to **2,300 sea turtles**. Table 8 summarizes the number of sea turtle takes anticipated per year based on the number of vessels.

Table 8. Anticipated Number of Sea Turtle Takes over 150 Years Based on Number of Vessels

Year	Number of vessels	Sea turtle takes
2019	4	368
2020	2	184
2021	4	368
2022	3	276
2023	3	276
2024	3	276
2025	3	276
2026	3	276
Total	**25**	**2,300**

5.1.1 Species take percentages

We used the 2007-2015 offshore STSSN data for North Carolina in order to determine the expected amount of take for each species. The 9-year dataset for North Carolina shows a total of 2,190 sea turtle strandings. Based on the artificial reef locations and substrate type, we believe this is the best available data to estimate the relative abundance of sea turtle species in the action areas and therefore, the percentages of sea turtle take by species as a result of the proposed action (Table 9).

Table 9. 2007 – 2015 North Carolina Sea Turtle Stranding Data

Species	2007	2008	2009	2010	2011	2012	2013	2014	2015	Total	Species % Comp
Loggerhead	113	117	150	220	119	99	116	101	85	1120	51.14155
Green	58	57	106	83	57	57	63	38	70	589	26.89498
Kemp's ridley	14	22	46	62	78	44	61	78	45	450	20.54795
Leatherback	4	4	3		6	3	2	3	5	30	1.369863
Hawksbill			1							1	0.045662
Grand Total	194	205	312	368	262	203	245	225	209	2190	100

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

Expected takes by species for artificial reefs over a 150-year time frame out of 2,300 anticipated sea turtles
= total expected sea turtle takes in 150 years from artificial reefs (2,300) x percent composition from stranding data for each species
Expected takes for loggerhead sea turtles over 150 years = 2,300 x 0.5114155 = 1,176.25565
Expected takes for green sea turtles over 150 years = 2,300 x 0.2689498 = 618.58454
Expected takes for Kemp's ridley sea turtles over 150 years = 2,300 x 0.2054795 = 472.60285
Expected takes for leatherback sea turtles over 150 years = 2,300 x 0.01369863 = 31.506849
Expected takes for hawksbill sea turtles over 150 years = 2,300 x 0.00045662 = 1.050226

Table 10. Breakdown of Species Based on Stranding Data

Species	Percent from stranding data	Species breakdown out of 2,300 anticipated sea turtle takes
Loggerhead	51.14155	1,176.25565
Green	26.89498	618.58454
Kemp's ridley	20.54795	472.60285
Leatherback	1.369863	31.506849
Hawksbill	0.045662	1.050226

North Atlantic and South Atlantic Green Sea Turtle DPSs

As described in Section 3.2.1b, information suggests that the vast majority of the anticipated green sea turtles caught in the Gulf of Mexico and South Atlantic regions are likely to come from the North Atlantic DPS. However, it is possible that animals from the South Atlantic DPS could be captured during the proposed action. We assume based on Bass and Witzell (2000) that 95% of animals affected by the proposed action are from the North Atlantic DPS and that 5% of the green sea turtles affected by the proposed action are from the South Atlantic DPS. Applying these percentages to our estimated take of 618.58454 green sea turtles over 150 years and rounding in such a way as to conservatively assume the most lethal captures, results in an estimated catch of up to 588 green sea turtles from the North Atlantic DPS (618.58454*0.95=587.655313, rounded up to 588), and an estimated catch of up to 31 green sea turtles from the South Atlantic DPS (618.58454*0.05= 30.929227, rounded up to 31). We note rounding when splitting the take into the two DPSs results in a slightly higher combined total (i.e., 619 instead of 618) than the 150-year actual estimate.

Table 11 summarizes the total number of anticipated lethal takes for each species of sea turtle. Totals from Table 10 were rounded up from the 10^{th} percentile to be most conservative for the species. We note rounding estimates from Table 11 results in a slightly higher total number of sea turtle takes (i.e., 2302 instead of 2300).

Table 11. Anticipated Amount of Lethal Take due to Artificial Reef Material Over a Period of 150 Years

Species	Lethal Take
Loggerhead sea turtle (NWA DPS)	1,177
Green sea turtle (NA DPS)	588
Green sea turtle (SA DPS)	31
Kemp's ridley sea turtle	473
Leatherback sea turtle	32
Hawksbill sea turtle	1
Total sea turtle take	**2,302**

6. CUMULATIVE EFFECTS

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

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

6.1. Vessel Interactions

NMFS's STSSN data indicate that vessel interactions are responsible for a large number of sea turtles stranding within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded sea turtles have obvious propeller or collision marks (Dwyer et al. 2003). Still, it is not always clear whether the collision occurred pre- or post-mortem. We believe that sea turtle injuries and mortalities by vessel interactions will continue in the future.

6.2. Pollution

Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from the pollution on sea turtles. However, the level of impacts cannot be projected. Marine debris (e.g., discarded fishing line or lines from boats) can entangle sea turtles in the water and drown them. Sea turtles commonly ingest plastic or mistake debris for food. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging behavior. As mentioned previously, sea turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for sea turtles and hinder their capability to forage, eventually they would tend to leave or avoid these areas (Ruben and Morreale 1999).

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

6.3. Global Climate Change

Global climate change is likely adversely affecting sea turtles. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events and fluctuation of precipitation levels, and change in air and water temperatures. There are multiple hypothesized effects to sea turtles including changes in their range and distribution as well as prey distribution and/or abundance due to water temperature changes. Ocean acidification may also negatively affect marine life, particularly organisms with calcium carbonate shells that serve as important prey items for many species. Global climate change may also affect reproductive behavior in sea turtles, including earlier onset of nesting, shorter intervals between nesting, and a decrease in the length of nesting season. Sea level rise may also reduce the amount of nesting beach available. Changes in air temperature may also affect the sex ratio of sea turtle hatchlings. A decline in reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of sea turtles in the Atlantic.

6.4. Coastal Development

Within the action area, beachfront development, lighting, and beach erosion potentially reduce or degrade sea turtle nesting habitats or interfere with hatchlings movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. Coastal counties are presently adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting. Some of these measures were drafted in response to lawsuits brought against the counties by concerned citizens who charged the counties with failing to uphold the ESA by allowing unregulated beach lighting that results in takes of hatchlings.

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

7. JEOPARDY ANALYSIS

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

It is the responsibility of the action agency to "insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered species or threatened species…" (ESA Section 7(a)(2)). Action agencies must consult with and seek assistance from the NMFS to meet this responsibility. NMFS must ultimately determine in a Biological Opinion whether the action jeopardizes listed species. To *jeopardize the continued existence of* is defined as "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). The following jeopardy analysis first considers the effects of the action to determine if we would reasonably expect the action to result in reductions in reproduction, numbers, or distribution of ESA-listed sea turtles. The analysis next considers whether any such reduction would in turn result in an appreciable reduction in the likelihood of survival of these species in the wild, and the likelihood of recovery of these species in the wild.

7.1. Loggerhead Sea Turtle (NWA DPS)

The proposed action may result in the lethal take of 1,177 loggerhead sea turtles from the NWA DPS over the next 150 years.

7.1.1 Survival

The lethal take of 1,177 loggerhead sea turtles over the next 150 years is a reduction in numbers. A lethal take could also result in a potential reduction in future reproduction, assuming the individual would be female and would have survived to reproduce in the future. For example, an adult female loggerhead sea turtle can lay approximately 4 clutches of eggs every 3 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. A reduction in the distribution of loggerhead sea turtles is not expected from lethal takes attributed to the proposed action. The anticipated lethal take is expected to occur in a discrete action area and loggerhead sea turtles in the NWA DPS generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether or not the reductions in loggerhead sea turtle numbers and reproduction attributed to the proposed action would appreciably reduce the likelihood of survival depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends (i.e., whether the estimated reductions, when viewed within the context of the environmental baseline and status of the species, are of such an extent that adverse effects on population dynamics are appreciable). In Section 3.3.1.1, we reviewed the status of this species in terms of nesting and female population trends and several recent assessments based on population modeling (i.e., (Conant et al. 2009; NMFS 2009). Below we synthesize what that information means both in general terms and the more specific context of the proposed action.

Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerhead sea turtles require high survival rates throughout their life to maintain a population. In other words, late-maturing species cannot tolerate much anthropogenic mortality without going into decline. Conant et al. (2009) concluded loggerhead natural growth rates are small, natural survival needs to be high, and even low- to moderate mortality can drive the population into decline. Because recruitment to the adult population is slow, population modeling studies suggest even small increased mortality rates in adults and subadults could substantially impact population numbers and viability (Chaloupka and Musick 1997; Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1995).

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

NMFS (2011b) preliminarily estimated the loggerhead population in the Northwestern Atlantic Ocean along the continental shelf of the Eastern Seaboard during the summer of 2010 at 588,439 individuals (estimate ranged from 381,941 to 817,023) based on positively identified individuals. The NMFS-NEFSC's point estimate increased to approximately 801,000 individuals when including data on unidentified sea turtles that were likely loggerheads. The NMFS-NEFSC

(2011) underestimates the total population of loggerheads since it did not include Florida's east coast south of Cape Canaveral or the Gulf of Mexico, which are areas where large numbers of loggerheads are also expected. In other words, it provides an estimate of a subset of the entire population.

Florida accounts for more than 90% of U.S. loggerhead nesting. The Florida Fish and Wildlife Conservation Commission conducted a detailed analysis of Florida's long-term loggerhead nesting data (1989-2017). They indicated that following a 24% increase in nesting between 1989 and 1998, nest counts declined sharply from 1999 to 2007. However, annual nest counts showed a strong increase (71%) from 2008 to 2016. Examining only the period between the high-count nesting season in 1998 and the 2016 nesting season, researchers found a slight but insignificant increase, indicating a reversal of the post-1998 decline. Nesting in 2017 declined relative to 2016, back to levels seen in 2013 and 2015. The overall change in counts from 1989 to 2017 was significantly positive; however, it should be noted that wide confidence intervals are associated with this complex data set (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/).

Abundance estimates accounting for only a subset of the entire loggerhead sea turtle population in the western North Atlantic indicate the population is large (i.e., several hundred thousand individuals). Nesting trends have been significantly increasing over several years against the background of the past and ongoing human and natural factors (as contemplated in the Status of the Species and Environmental Baseline) that have contributed to the current status of the species. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record.

The proposed action could lethally take 1,177 loggerhead sea turtles over the next 150 years. We do not expect this loss to result in a detectable change to the population numbers or increasing trends because this loss 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. 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 2011b). After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the loggerhead sea turtle NWA DPS in the wild.

7.1.2 Recovery

The loggerhead recovery plan 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. 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 150 year time period, as a result of the proposed action is not expected to be detectable on a population level and therefore not expected to affect population growth over the timeframe analyzed.

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. As noted previously, we believe the future takes predicted will not cause a detectible increase to the current levels of anthropogenic take that has occurred in the past and those past takes did not impede the positive trends we are currently seeing in nesting during that time. We also indicated that the lethal take of 1,177 loggerhead sea turtles over the next 150 years is so small in relation to the overall population, that it would not impede achieving the Recovery Objectives, even when considered in the context of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. We believe this is true for both nesting and juvenile in-water populations. For these reasons, we do not believe the proposed action will impede achieving the recovery objectives or overall recovery strategy.

7.1.3 Conclusion

The lethal take of loggerhead sea turtles associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NWA DPS of the loggerhead sea turtle in the wild.

7.2. Green Sea Turtle (NA and SA DPSs)

As discussed in the effects of the action section, within U.S. waters green sea turtles from both the NA and SA DPSs can be found on foraging grounds. While there are currently no in-depth studies available to determine the percent of NA and SA DPS individuals in any given location, an analysis of green sea turtles on the foraging grounds off Hutchinson Island, Florida (Atlantic Ocean-side), found approximately 95% of the turtles sampled came from the NA DPS. While it is highly likely green sea turtles found in or near the action area will be from the NA DPS, we cannot rule out that they may also be from the SA DPS. Therefore, to analyze effects in a precautionary manner, we will conduct 2 jeopardy analyses, one for each DPS (i.e., assuming up to 95% could come from the NA DPS and 5% could come from the SA DPS).

7.2.1 NA DPS

The proposed action may result in the lethal take of 588 green sea turtles from the NA DPS over the next 150 years.

7.2.1.1 <u>Survival</u>

The lethal take of 588 green sea turtles from the NA DPS over the next 150 years as a result of the proposed action is a reduction in numbers. A lethal take could also result in a potential reduction in future reproduction, assuming the individual would be 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 large action area (i.e., where the deployment of artificial reef material is occurring); however, the size of each reef compared to the action area as a whole is small and patchy, and green sea turtle generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

Seminoff et al. (2015) estimated that there are greater than 167,000 nesting green sea turtle females in the NA DPS. The nesting at Tortuguero, Costa Rica, accounts for approximately 79% of that estimate (approximately 131,000 nesters), with Quintana Roo, Mexico, (approximately 18,250 nesters; 11%), and Florida, USA (approximately 8,400 nesters; 5%), also accounting for a large portion of the overall nesting (Seminoff et al. 2015). At Tortuguero, Costa Rica, the number of nests laid per year from 1999 to 2010 increased, despite substantial human impacts to the population at the nesting beach and at foraging areas (Campell and Lagueux 2005; Troëng 1998; Troëng and Rankin 2005). Nesting locations in Mexico along the Yucatan Peninsula also indicate the number of nests laid each year has deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS 2007a). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpubl. data, 2013, in Seminoff et al. 2015). In Florida, most nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, FFWCC, pers. comm., 2013). As described in the Section 3, nesting has increased substantially over the last 20 years and peaked in 2017 with 38,954 nests statewide.

In summary, green sea turtle nesting at the primary nesting beaches within the range of the NA DPS has been increasing over the past 2 decades, against the background of the past and ongoing human and natural factors (i.e., the environmental baseline) that have contributed to the current status of the species. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Our evaluation of the impact of future captures is based in part on our belief that the same level of capture occurred in the past, yet we have still seen positive trends in the status of this species. Since the abundance trend information for NA DPS green sea turtles is clearly increasing, we believe the potential lethal take of 588 green sea turtles from the NA DPS over the next 150 years will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle NA DPS in the wild.

7.2.1.2 Recovery

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

> *Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.*
>
> *Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.*

According to data collected from Florida's index nesting beach survey from 1989-2017, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 38,954 in 2017, indicating that the first listed recovery objective is currently being met. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting, however, it is likely that numbers on foraging grounds have also increased, consistent with the criteria of the second listed recovery objective.

The potential lethal take of up to 588 green sea turtles from the NA DPS over the next 150 years as a result of the proposed action will result in a reduction in numbers when it occurs. Our evaluation of potential future mortality is based our belief that the same level of interactions occurred in the past, and with that level we have still seen positive trends in the status of this species. This take is unlikely to have any detectable influence on the recovery objectives and trends noted above, and will not result in an appreciable reduction in the likelihood of NA DPS green sea turtles' recovery in the wild even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion.

7.2.1.3 Conclusion

The lethal take of 588 green sea turtles from the NA DPS associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NA DPS of green sea turtle in the wild.

7.2.2 SA DPS

The proposed action may result in the lethal take of 31 green sea turtle from the SA DPS over the 150 years as a result of the artificial reefs (618.58454 total lethal takes*0.05= 30.929227, rounded up to 31).

7.2.2.1 Survival

The lethal take of 31 green sea turtle from the SA DPS from artificial reefs is a reduction in numbers. As discussed above, lethal interactions would also result in a potential reduction in future reproduction, assuming the individual would be female and would have survived otherwise to reproduce. The anticipated lethal takes are expected to occur over a long time period (150 years) and over a large action area (i.e., where the deployment of artificial reef material is occurring); however, the size of each reef compared to the action area as a whole is small and patchy, and green sea turtles in the SA DPS generally have large ranges; thus, no reduction in their distribution is expected from the take of these individuals.

Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

In Section 4.2, we summarized available information on number of nesters and nesting trends at SA DPS beaches. Seminoff et al. (2015) estimated that there are greater than 63,000 nesting females in the SA DPS, though they noted the adult female nesting abundance from 37 beaches could not be quantified. The nesting at Poilão, Guinea-Bissau, accounted for approximately 46% of that estimate (approximately 30,000 nesters), with Ascension Island, United Kingdom, (approximately 13,400 nesters; 21%), and the Galibi Reserve, Suriname (approximately 9,400 nesters; 15%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015).

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

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

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

Our evaluation of the impact of future captures is based in part on our belief that the same level of capture occurred in the past, yet we have still seen positive trends in the status of this species. In summary, nesting at the primary nesting beaches for the SA DPS has been increasing over the past 3 decades, against the background of the past and ongoing human and natural factors (as contemplated in the Status of the Species and Environmental Baseline sections) that have contributed to the current status of the species. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for green sea turtles is clearly increasing, we believe the potential lethal take of 31 green sea turtle from the SA DPS over the next 150 years as a result of the project will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle SA DPS in the wild.

7.2.2.2 Recovery

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

> *Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.*
>
> *Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.*

The nesting recovery objective is specific to the NA DPS, but demonstrates the importance of increases in nesting to recovery. As previously stated, nesting at the primary SA DPS nesting beaches has been increasing over the past 3 decades. Our evaluation of potential future mortality is based our belief that the same level of interactions occurred in the past, and with that level we have still seen positive trends in the status of this species. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting and in-water abundance, however, it is likely that numbers on foraging grounds have increased.

The potential lethal take of 31 green sea turtle from the SA DPS over the next 150 years as a result of the project will result in a reduction in numbers when it occurs, but it is unlikely to have any detectable influence on the trends noted above, even when considered in context with the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. Thus, the proposed action will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of the SA DPS of green sea turtles' recovery in the wild.

7.2.2.3 Conclusion

The potential lethal take of 31 SA DPS green sea turtles associated with the proposed action is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of green sea turtle in the wild.

7.3. Kemp's ridley Sea Turtle

The proposed action may result in the lethal take of 473 Kemp's ridley sea turtles over the next 150 years.

7.3.1 Survival

The potential lethal take of up to 473 Kemp's ridley sea turtle over the next 150 years as a result of the proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. The Turtle Expert Working Group (Turtle Expert Working Group 1998) estimates age at maturity from 7-15 years. Females return to their nesting beach about every 2 years (Turtle Expert Working Group 1998). The mean clutch size for Kemp's ridley sea turtle is 100 eggs/nest, with an average of 2.5 nests/female/season. Lethal takes could also result in a potential reduction in future reproduction, assuming at least one of these individuals would be female and would have survived to reproduce in the future. The loss of 473 Kemp's ridley sea turtle 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 large action area (i.e., where the deployment of artificial reef material is occurring); however, the size of each reef compared to the action area as a whole is small and patchy, and Kemp's ridley sea turtle generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

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).

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, http://www.nps.gov/pais/naturescience/strp.htm, http://www.nps.gov/pais/naturescience/current-season.htm). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015-2017.

It is important to remember that with significant inter-annual variation in nesting data, sea turtle population trends necessarily are measured over decades and the long-term trend line better reflects the population increase in Kemp's ridleys. With the recent increase in nesting data (2015-16) and recent declining numbers of nesting females (2013-14), it is too early to tell whether the long-term trend line is affected. Nonetheless, long-term data from 1990 to present continue to support that Kemp's ridley sea turtle is increasing in population size.

Our evaluation of the impact of future captures is based in part on our belief that the same level of capture occurred in the past, yet we have still seen positive trends in the status of this species. We believe this long-term increasing trend in nesting is evidence of an increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information is clearly increasing, we believe the potential lethal take of 473 Kemp's ridley sea turtles over the next 150 years attributed to the proposed action will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of Kemp's ridley sea turtles in the wild.

7.3.2 Recovery

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

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

With respect to this recovery objective, the nesting numbers in 2018 indicate there were a total of 17,945 nests on the main nesting beaches in Mexico. This number represents approximately 7,178 nesting females for the season based on 2.5 clutches/female/season. The number of nests reported annually from 2010 to 2014 overall declined; however they rebounded in 2015 through 2017. 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 473 Kemp's ridley sea turtles over the next 150 years is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede the progress toward achieving this recovery objective. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of Kemp's ridley sea turtles' recovery in the wild.

7.3.3 Conclusion

The lethal take of 473 Kemp's ridley sea turtles associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of Kemp's ridley sea turtle in the wild.

7.4. Leatherback Sea Turtle

The proposed action may result in up to 32 leatherback sea turtle lethal takes. The anticipated lethal takes are expected to occur over a long time period (150 years) and over a large action area (i.e., where the deployment of artificial reef material is occurring); however, the size of each reef compared to the action area as a whole is small and patchy, and leatherback sea turtles generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

7.4.1 Survival

The lethal take of up to 32 leatherback sea turtles over the next 150 years as a result of the project would reduce the population. 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 Leatherback Turtle Expert Working Group estimated there are between 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic based on 2004 and 2005 nesting count data (Turtle Expert Working Group 2007). The potential loss of up to 32 leatherback sea turtles over the next 150 years accounts for only 0.0003-0.001% of those population estimates, which are only a subset of the entire population. We do not believe these potential loses will have any detectable impact on these population numbers.

Of the 15 leatherback nesting populations in the North Atlantic, 7 show an increase in nesting (Florida, Puerto Rico [not Culebra], St. Croix-U.S. Virgin Islands, British Virgin Islands, Trinidad, Guyana, and Brazil) and 3 have shown a decline in nesting (Puerto Rico [Culebra], Costa Rica [Tortuguero], and Costa Rica [Gandoca]). The most important nesting populations (French Guiana and Suriname) have remained stable. Suriname and French Guiana may represent over 40% of the world's leatherback nesting population (Spotila et al. 1996), accounting for between 31,000 to 60,000 nests annually (NMFS and USFWS 2013b).

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

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

Our evaluation of the impact of future captures is based in part on our belief that the same level of capture occurred in the past, yet we have still seen positive trends in the status of this species. Aside from the nesting declines in Tortuguero, which are significant, most of the other nesting populations appear to be increasing or are remaining stable, including the most significant populations in French Guiana and Suriname. Since we anticipate a low number of mortalities over the next 150 years and we have no reason to believe nesting females will be disproportionately affected, we believe the potential mortalities associated with the proposed action will have no detectable effect on current nesting trends.

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

7.4.2 Recovery

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

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

We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. As noted previously, the Florida and St. Croix nesting populations are increasing. The nesting population in Culebra, Puerto Rico, had been increasing since the late 1970s but has been declining in recent years; however, it appears these declines may reflect a shift in nest site fidelity rather than a decline in the female population. Since we concluded that the potential loss of up to 32 leatherback sea turtles over the next 150 years is not likely to have any detectable effect on these nesting trends, we do not believe the proposed action is impeding the progress toward achieving this recovery objective. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild.

7.4.3 Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the leatherback sea turtle in the wild.

7.5. Hawksbill Sea Turtle

The proposed action may result in up to 1 lethal hawksbill sea turtle take over the next 150 years. The anticipated lethal take is expected to occur over a large action area (i.e., where the deployment of artificial reef material is occurring); however, the size of each reef compared to the action area as a whole is small and patchy, and hawksbill sea turtles generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

7.5.1 Survival

The lethal take of up to 1 hawksbill sea turtle over the next 150 years as a result of the project would reduce the number of hawksbill sea turtles, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Any potential lethal take could also result in a reduction in future reproduction, assuming the individual would be a female and would have survived to reproduce in the future. For example, an adult hawksbill sea turtle can lay 3-5 clutches of eggs every few years (Meylan and Donnelly 1999; Richardson et al. 1999) with up to 250 eggs/nest (Hirth and Latif 1980). Thus, the loss of a female could preclude the production of thousands of eggs and hatchlings, of which a fraction would otherwise survive to sexual maturity and contribute to future generations.

In the absence of any total population estimates for hawksbill sea turtles, nesting trends are the best proxy we have for estimating population changes. The most recent 5-year status review estimated between 22,000 and 29,000 adult females existed in the Atlantic basin in 2007 (NMFS and USFWS 2013a); this estimate does not include juveniles of either sex or mature males. Hawksbill nesting trends also indicate an improvement over the last 20 years. A survey of historical nesting trends (i.e., 20-100 years ago) for the 33 nesting sites in the Atlantic Basin found declines at 25 of those sites and data were not available for the remaining 8 sites.

However, in the last 20 years, nesting trends have been improving. Of those 33 sites, 10 sites now show an increase in nesting, 10 sites showed a decrease, and data for the remaining 13 are not available (NMFS and USFWS 2013a).

Our evaluation of the impact of future captures is based in part on our belief that the same level of capture occurred in the past, yet we have still seen positive trends in the status of this species. We believe increases in nesting over the last 20 years, relative to the historical trends, indicate improving population numbers. Additionally, even when we conservatively evaluate the potential effects of the proposed action on a portion of the hawksbill population (i.e., adult females) we believe the impacts will be minor relative to the entire population. Thus, we believe the potential loss of up to 1 hawksbill sea turtle over the next 150 years will not have any detectable effect on the population, distribution or reproduction of hawksbills. Therefore, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

7.5.2 Recovery

The Recovery Plan for the population of the hawksbill sea turtles (NMFS and USFWS 1993) lists the following relevant recovery objectives over a period of 25 continuous years:

> *Objective: The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests on at least 5 index beaches, including Mona Island (Puerto Rico) and Buck Island Reef National Monument (U.S. Virgin Islands).*
>
> *Objective: The numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least 5 key foraging areas within Puerto Rico, USVI, and Florida.*

Although the most recent 5-year review indicates there is not enough information to evaluate the statistical significance of nesting trends, nesting populations are increasing at the Puerto Rico (Mona Island) and U.S. Virgin Islands (Buck Island Reef National Monument) index beaches. Also in the U.S. Caribbean, additional nesting beaches are now being more systematically monitored to allow for future population trend assessments. Elsewhere in the Caribbean outside U.S. jurisdiction, nesting populations in Antigua/Barbuda and Barbados are increasing; however, other important nesting concentrations in the insular Caribbean are decreasing or their status is unknown, including Antiqua/Barbuda (except Jumby Bay), Bahamas, Cuba (Doce Leguas Cays), Jamaica, and Trinidad and Tobago (NMFS and USFWS 2013a). Based on this information we do not expect the loss of 1 hawksbill sea turtle over the next 150 years will impede this recovery objective.

The proposed action could cause the loss of up to 1 hawksbill sea turtles over the next 150 years and the animals may or may not be adult and may or may not be female. Our evaluation of potential future mortality is based our belief that the same level of interactions occurred in the past, and with that level we have still seen positive trends in the status of this species. We determined the potential lethal captures associated with the proposed action would not have any detectable influence on the magnitude of those trends. While information on trends for adults, subadults, and juveniles at key foraging areas is not yet available, we also believe it is unlikely the potential removal of 1 hawksbill over the next 150 years will have any detectable influence over the numbers of adults, subadults, and juveniles occurring at 5 key foraging areas. Thus, we believe the proposed action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild.

7.5.3 Conclusion

The effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the hawksbill sea turtle in the wild.

8. CONCLUSION

Using the best available data, we analyzed the effects of the proposed action in the context of the status of the species, the environmental baseline, and cumulative effects and we determined that the proposed action is not likely to jeopardize the continued existence of green sea turtle (NA or SA DPS), loggerhead sea turtle (NWA DPS), Kemp's ridley sea turtle, leatherback sea turtle, and hawksbill sea turtle. These analyses focused on the impacts to, and population responses of, these species. Because the proposed action will not appreciably reduce the likelihood of survival and recovery of these species, it is our Opinion that the proposed action is also not likely to jeopardize the continued existence of any of these species.

9. INCIDENTAL TAKE STATEMENT

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

NMFS anticipates the total lethal take over the next 150 years as a result of the project will consist of up to 1,177 loggerhead sea turtles (NWA DPS), 588 green sea turtles (NA DPS), 31 green sea turtles (South Atlantic DPS), 473 Kemp's ridley sea turtles, 32 leatherback sea turtles, and 1 hawksbill sea turtle (Table 12). Based on the best available data, we do not anticipate any non-lethal take of the species listed above. The level of takes occurring annually is highly variable and influenced by sea temperatures, species abundances, monofilament accumulation, and other factors that cannot be predicted.

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

Sea Turtles	Total Estimated Lethal Take over 150 Years
Loggerhead	1,177
N. Atlantic Green DPS	588
S. Atlantic Green DPS	31
Kemp's ridley	473
Leatherback	32
Hawksbill	1

If any takes of species under NMFS's purview are observed taken during the proposed action authorized using this Opinion as the Section 7 consultation, it shall be immediately reported to takereport.nmfsser@noaa.gov (include Opinion issue date, and the NMFS PCTS ECO identifier number [SER-2019-19783/SERO-2019-00225]).

The USACE has a continuing duty to regulate the activity covered by this ITS. If the USACE (1) fails to assume and implement the T&Cs or (2) fails to require the T&Cs of the ITS through enforceable terms that are added to the permit or grant 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 ITS (50 CFR §402.14(i)(3)).

9.1. Effect of the Take

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

10. REASONABLE AND PRUDENT MEASURES

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

The RPMs and T&Cs are specified as required, by 50 CFR 402.01(i)(1)(ii) and (iv), to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. These RPMs and T&Cs are nondiscretionary, and must be implemented by the USACE in order for the protection of Section 7(o)(2) to apply. The USACE has a continuing duty to regulate the activity covered by this ITS. If the USACE fails to adhere to the T&Cs through enforceable terms, and/or fails to retain oversight to ensure compliance with these T&Cs, the protective coverage of Section 7(o)(2) may lapse.

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

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

11. TERMS AND CONDITIONS

The following T&Cs include the following:

1) If the Permittee discovers or observes any live, damaged, injured or dead individual of an endangered or threatened species during construction, the Permittee shall immediately notify the Wilmington District Engineer so that required coordination can be initiated with the U.S. Fish and Wildlife Service and/or National Marine Fisheries Service.

2) The Permittee shall report any injured or dead turtles observed during the annual site monitoring events.

3) The USACE must submit an annual report as described in Section 2.4 of this Opinion.

12. CONSERVATION RECOMMENDATIONS

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

1) Artificial reef programs should include a qualitative assessment of monofilament accumulation on artificial reef structure during artificial reef monitoring dives.

2) All vessels with a planned deployment depth below recognized recreational diving depths (i.e., 130-150 ft) should remove all railing and other non-essential structure that could otherwise easily accumulate monofilament line.

3) The Permittee should require the placement and maintenance of acoustic telemetry receivers at each artificial reef site.

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

13. REINITIATION OF CONSULTATION

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

14. LITERATURE CITED

40th Northeast Regional Stock Assessment Workshop (40th SAW). 2005. 40th SAW assessment report. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Reference Document 05-04, Woods Hole, MA.

Addison, D. S. 1997. Sea turtle nesting on Cay Sal, Bahamas, recorded June 2-4, 1996. Bahamas Journal of Science 5(1):34-35.

Addison, D. S., and B. Morford. 1996. Sea turtle nesting activity on the Cay Sal Bank, Bahamas. Bahamas Journal of Science 3(3):31-36.

Aguirre, A. A., G. H. Balazs, T. R. Spraker, S. K. K. Murakawa, and B. Zimmerman. 2002. Pathology of oropharyngeal fibropapillomatosis in green turtles *Chelonia mydas*. Journal of Aquatic Animal Health 14:298-304.

Aguirre, A. A., G. H. Balazs, B. Zimmerman, and F. D. Galey. 1994. Organic contaminants and trace metals in the tissues of green turtles (*Chelonia mydas*) afflicted with fibropapillomas in the Hawaiian Islands. Marine Pollution Bulletin 28(2):109-114.

Amos, A. F. 1989. The occurrence of hawksbills (*Eretmochelys imbricata*) along the Texas coast. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFC-232, Miami, FL.

Arendt, M. D., and coauthors. 2009. Examination of local movement and migratory behavior of sea turtles during spring and summer along the Atlantic Coast off the Southeastern United States. South Carolina Department of Natural Resources, Grant Number NA03NMF4720281.

Arendt, M. D., and coauthors. 2012. Migration, distribution, and diving behavior of adult male loggerhead sea turtles (*Caretta caretta*) following dispersal from a major breeding aggregation in the Western North Atlantic. Marine Biology 159(1).

Atlantic Sturgeon Status Review Team. 2007. Status review of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

Avens, L., J. C. Taylor, L. R. Goshe, T. T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8:165-177.

Backus, R. H., S. Springer, and E. L. Arnold. 1956. A contribution to the natural history of the white-tip shark, *Pterolamiops longimanus* (Poey). Deep Sea Research (1953) 3(3):178-188.

Balazs, G. H. 1979. Growth rates of immature green turtles in the Hawaiian Archipelago. Smithsonian Institution Press, Washington, D.C.

Balazs, G. H. 1983. Recovery records of adult green turtles observed or originally tagged at French Frigate Shoals, northwestern Hawaiian Islands. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center, NOAA Technical Memorandum NMFS-SWFC-36 and UNIHI-SEAGRANT-CR-83-03, Honolulu, HI.

Balazs, G. H. 1985. Impact of ocean debris on marine turtles: Entanglement and ingestion. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center, Technical Memorandum NMFS-SWFC-54, Honolulu, HI.

Barnette, M. C. 2017. Potential impacts of artificial reef development on sea turtle conservation in Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, NOAA Technical Memorandum NMFS-SER-5, Saint Petersburg, FL.

Bass, A. L., and coauthors. 1996. Testing models of female reproductive migratory behaviour and population structure in the Caribbean hawksbill turtle, *Eretmochelys imbricata*, with mtDNA sequences. Molecular Ecology 5:321-328.

Bass, A. L., and W. N. Witzell. 2000. Demographic composition of immature green turtles (*Chelonia mydas*) from the east central Florida coast: Evidence from mtDNA markers. Herpetologica 56(3):357-367.

Benson, S. R., and coauthors. 2007a. Post-nesting migrations of leatherback turtles (*Dermochelys coriacea*) from Jamursba-Medi, Bird's Head Peninsula, Indonesia. Chelonian Conservation and Biology 6(1):150-154.

Benson, S. R., and coauthors. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere 2(7):27.

Benson, S. R., K. A. Forney, J. T. Harvey, J. V. Carretta, and P. H. Dutton. 2007b. Abundance, distribution, and habitat of leatherback turtles (*Dermochelys coriacea*) off California, 1990−2003. Fishery Bulletin 105(3):337-347.

Bjorndal, K. A. 1982. The consequences of herbivory for the life history pattern of the Caribbean green turtle, *Chelonia mydas*. Pages 111-116 *in* K. A. Bjorndal, editor. Biology and

Conservation of Sea Turtles: Proceedings of the World Conference on Sea Turtle Conservation. Smithsonian Institution Press, Washington D.C.

Bjorndal, K. A. 1997. Foraging ecology and nutrition of sea turtles. Pages 199–231 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Bjorndal, K. A., and A. B. Bolten. 2000. Proceedings of a Workshop on Assessing Abundance and Trends for In-water Sea Turtle Populations. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-445, Miami, FL.

Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. 2005. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. Ecological Applications 15(1):304-314.

Bjorndal, K. A., A. B. Bolten, T. Dellinger, C. Delgado, and H. R. Martins. 2003. Compensatory growth in oceanic loggerhead sea turtles: Response to a stochastic environment. Ecology 84(5):1237-1249.

Bjorndal, K. A., J. A. Wetherall, A. B. Bolten, and J. A. Mortimer. 1999. Twenty-six years of green turtle nesting at Tortuguero, Costa-Rica: An encouraging trend. Conservation Biology 13(1):126-134.

Bolten, A. B., and coauthors. 1998. Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis. Ecological Applications 8(1):1-7.

Bolten, A. B., and B. E. Witherington. 2003. Loggerhead Sea Turtles. Smithsonian Books, Washington, D.C.

Bonfil, R., S. Clarke, and H. Nakano. 2009. The biology and ecology of the oceanic whitetip shark, *Carcharhinus longimanus*. Pages 128-139 *in* T. J. Pitcher, M. D. Camhi, E. K. Pikitch, and E. A. Babcock, editors. Sharks of the Open Ocean: Biology, Fisheries and Conservation. Blackwell Publishing.

Bostrom, B. L., and D. R. Jones. 2007. Exercise warms adult leatherback turtles. Comparative Biochemistry and Physiology A: Molecular and Integrated Physiology 147(2):323-31.

Boulon Jr., R. H. 1983. Some notes on the population biology of green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtles in the northern U.S. Virgin Islands: 1981-1983. Government of the Virgin Islands, Division of Fish and Wildlife, Grant No. NA82-GA-A-00044, St. Thomas, U.S. Virgin Islands.

Boulon Jr., R. H. 1994. Growth rates of wild juvenile hawksbill turtles, *Eretmochelys imbricata*, in St. Thomas, United States Virgin Islands. Copeia 1994(3):811-814.

Bowen, B. W., and coauthors. 1992. Global population structure and natural history of the green turtle (*Chelonia mydas*) in terms of matriarchal phylogeny. Evolution 46(4):865-881.

Bowen, J. L., and I. Valiela. 2001. The ecological effects of urbanization of coastal watersheds: Historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries. Canadian Journal of Fisheries and Aquatic Sciences 58(8):1489-1500.

Bowlby, C. E., G. A. Green, and M. L. Bonnell. 1994. Observations of leatherback turtles offshore of Washington and Oregon. Northwestern Naturalist 75(1):33-35.

Bräutigam, A., and K. L. Eckert. 2006. Turning the tide: Exploitation, trade and management of marine turtles in the Lesser Antilles, Central America, Columbia and Venezuela. TRAFFIC International, Cambridge, United Kingdom.

Bresette, M. J., R. A. Scarpino, D. A. Singewald, and E. P. de Maye. 2006. Recruitment of post-pelagic green turtles (*Chelonia mydas*) to nearshore reefs on Florida's southeast coast. Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Broderick, A. C., and coauthors. 2006. Are green turtles globally endangered? Global Ecology and Biogeography 15(1):21-26.

Burchfield, P. M. 2013. Gladys Porter Zoo's preliminary annual report on the Mexico/United States of America population restoration project for the Kemp’s ridley sea turtle, *Lepidochelys kempii*, on the coasts of Tamaulipas, Mexico 2013. Gladys Porter Zoo, Brownsville, TX.

Burgess, K. B., and coauthors. 2016. Manta birostris, predator of the deep? Insight into the diet of the giant manta ray through stable isotope analysis. R Soc Open Sci 3(11).

Caldwell, D. K., and A. Carr. 1957. Status of the sea turtle fishery in Florida. Wildlife Management Institute, Washington, D.C.

Campell, C. L., and C. J. Lagueux. 2005. Survival probability estimates for large juvenile and adult green turtles (*Chelonia mydas*) exposed to an artisanal marine turtle fishery in the western Caribbean. Herpetologica 61(2):91-103.

Carballo, J. L., C. Olabarria, and T. G. Osuna. 2002. Analysis of four macroalgal assemblages along the Pacific Mexican coast during and after the 1997-98 El Niño. Ecosystems 5(8):749-760.

Carillo, E., G. J. W. Webb, and S. C. Manolis. 1999. Hawksbill turtles (*Eretmochelys imbricata*) in Cuba: An assessment of the historical harvest and its impacts. Chelonian Conservation and Biology 3(2):264-280.

Carr, A. 1986. New perspectives on the pelagic stage of sea turtle development. Caribbean Conservation Corporation and U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center, Panama City Laboratory, NOAA Technical Memorandum NMFS-SEFSC-190, Panama City, FL.

Carr, T., and N. Carr. 1991. Surveys of the sea turtles of Angola. Biological Conservation 58(1):19-29.

Catry, P., and coauthors. 2002. First census of the green turtle at Poilão, Bijagós Archipelago, Guinea-Bissau: The most important nesting colony on the Atlantic coast of Africa. Oryx 36(4):400-403.

Catry, P., and coauthors. 2009. Status, ecology, and conservation of sea turtles in Guinea-Bissau. Chelonian Conservation and Biology 8(2):150-160.

Caurant, F., P. Bustamante, M. Bordes, and P. Miramand. 1999. Bioaccumulation of cadmium, copper and zinc in some tissues of three species of marine turtles stranded along the French Atlantic coasts. Marine Pollution Bulletin 38(12):1085-1091.

Cetacean and Turtle Assessment Program. 1982. A characterization of marine mammals and turtles in the mid- and north Atlantic areas of the U.S. Outer Continental Shelf. Cetacean and Turtle Assessment Program (CeTAP), University of Rhode Island, Contract AA551-CT8-48, Report No. BLM/YL/TR-82/03, Kingston, RI.

Chacón-Chaverri, D., and K. L. Eckert. 2007. Leatherback sea turtle nesting at Gandoca Beach in Caribbean Costa Rica: Management recommendations from fifteen years of conservation. Chelonian Conservation and Biology 6(1):101-110.

Chaloupka, M. 2002. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. Ecological Modelling 148(1):79-109.

Chaloupka, M. Y., and C. J. Limpus. 1997. Robust statistical modeling of hawksbill sea turtle growth rates (southern Great Barrier Reef). Marine Ecology Progress Series 146: 1-8.

Chaloupka, M. Y., and C. J. Limpus. 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Marine Biology 146(6):1251-1261.

Chaloupka, M. Y., C. J. Limpus, and J. Miller. 2004. Green turtle somatic growth dynamics in a spatially disjunct Great Barrier Reef metapopulation. Coral Reefs 23(3):325-335.

Chaloupka, M. Y., and J. A. Musick. 1997. Age, growth, and population dynamics. Pages 233-276 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Chaloupka, M. Y., T. M. Work, G. H. Balazs, S. K. K. Murakawa, and R. Morris. 2008. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982-2003). Marine Biology 154(5):887-898.

Colburn, T., D. Dumanoski, and J. P. Myers. 1996. Our Stolen Future: Are We Threatening Our Fertility, Intelligence, and Survival? – A Scientific Detective Story. Dutton/Penguin Books, New York.

Conant, T. A., and coauthors. 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the Loggerhead Biological Review Team to the National Marine Fisheries Service, August 2009.

Corsolini, S., S. Aurigi, and S. Focardi. 2000. Presence of polychlorobiphenyls (PCBs) and coplanar congeners in the tissues of the Mediterranean loggerhead turtle *Caretta caretta*. Marine Pollution Bulletin 40(11):952-960.

Couturier, L. I. E., and coauthors. 2013. Stable isotope and signature fatty acid analyses suggest reef manta rays feed on demersal zooplankton. PLOS ONE 8(10):e77152.

Crabbe, M. J. C. 2008. Climate change, global warming and coral reefs: Modelling the effects of temperature. Computational Biology and Chemistry 32(5):311-314.

Crouse, D. T. 1999. Population modeling and implications for Caribbean hawksbill sea turtle management. Chelonian Conservation and Biology 3(2):185-188.

Crouse, D. T., L. B. Crowder, and H. Caswell. 1987. A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology 68(5):1412-1423.

Crowder, L. B., D. T. Crouse, S. S. Heppell, and T. H. Martin. 1994. Predicting the impact of turtle excluder devices on loggerhead sea turtle populations. Ecological Applications 4(3):437-445.

D'Ilio, S., D. Mattei, M. F. Blasi, A. Alimonti, and S. Bogialli. 2011. The occurrence of chemical elements and POPs in loggerhead turtles (*Caretta caretta*): An overview. Marine Pollution Bulletin 62(8):1606-1615.

Davenport, J., D. L. Holland, and J. East. 1990. Thermal and biochemical characteristics of the lipids of the leatherback turtle (*Dermochelys coriacea*): Evidence of endothermy. Journal of the Marine Biological Association of the United Kingdom 70:33-41.

Deepwater Horizon Natural Resource Damage Assessment Trustees. 2016. Deepwater Horizon oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

Diez, C. E., and R. P. van Dam. 2002. Habitat effect on hawksbill turtle growth rates on feeding grounds at Mona and Monito Islands, Puerto Rico. Marine Ecology Progress Series 234:301-309.

Diez, C. E., and R. P. van Dam. 2007. In-water surveys for marine turtles at foraging grounds of Culebra Archipelago, Puerto Rico - Progress Report: FY 2006-2007.

Dodd Jr., C. K. 1988. Synopsis of the biological data on the loggerhead sea turtle: *Caretta caretta* (Linnaeus 1758). U.S. Department of the Interior, Fish and Wildlife Service, FAO Synopsis NMFS-149, Gainesville, FL.

Doughty, R. W. 1984. Sea turtles in Texas: A forgotten commerce. The Southwestern Historical Quarterly 88(1):43-70.

Duque, V. M., V. P. Páez, and J. A. Patino. 2000. Ecología de anidación y conservación de la tortuga caná, *Dermochelys coriacea*, en la Playona, Golfo de Urabá Chocoano (Colombia), en 1998 Actualidades Biologicas Medellín 22(72):37-53.

Dutton, D. L., P. H. Dutton, M. Chaloupka, and R. H. Boulon. 2005. Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biological Conservation 126(2):186-194.

Dwyer, K. L., C. E. Ryder, and R. Prescott. 2003. Anthropogenic mortality of leatherback turtles in Massachusetts waters. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Eckert, K. L. 1995. Hawksbill sea turtle, *Eretmochelys imbricata*. Pages 76-108 *in* P. T. Plotkin, editor. Status Reviews for Sea Turtles Listed under the Endangered Species Act of 1973. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S Department of the Interior, Fish and Wildlife Service, Silver Spring, MD.

Eckert, K. L., and S. A. Eckert. 1990. Embryo mortality and hatch success in (*in situ*) and translocated leatherback sea turtle (*Dermochelys coriacea*) eggs. Biological Conservation 53:37-46.

Eckert, K. L., S. A. Eckert, T. W. Adams, and A. D. Tucker. 1989a. Inter-nesting migrations by leatherback sea turtles (*Dermochelys coriacea*) in the West Indies. Herpetologica 45(2):190-194.

Eckert, K. L., J. A. Overing, and B. B. Lettsome. 1992. WIDECAST sea turtle recovery action plan for the British Virgin Islands. UNEP Caribbean Environment Programme, Wider Caribbean Sea Turtle Recovery Team and Conservation Network, CEP Technical Report No. 15, Kingston, Jamaica.

Eckert, K. L., B. P. Wallace, J. G. Frazier, S. A. Eckert, and P. C. H. Pritchard. 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of the Interior, Fish and Wildlife Service, BTP-R4015-2012, Washington, D.C.

Eckert, S. A. 2006. High-use oceanic areas for Atlantic leatherback sea turtles (*Dermochelys coriacea*) as identified using satellite telemetered location and dive information. Marine Biology 149(5):1257-1267.

Eckert, S. A., and coauthors. 2006. Internesting and postnesting movements and foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. Chelonian Conservation and Biology 5(2):239-248.

Eckert, S. A., K. L. Eckert, P. Ponganis, and G. L. Kooyman. 1989b. Diving and foraging behavior of leatherback sea turtles (*Dermochelys coriacea*). Canadian Journal of Zoology 67(11):2834-2840.

Eckert, S. A., D. W. Nellis, K. L. Eckert, and G. L. Kooyman. 1984. Deep diving record for leatherbacks. Marine Turtle Newsletter 31:4.

Eckert, S. A., and L. Sarti. 1997. Distant fisheries implicated in the loss of the world's largest leatherback nesting population. Marine Turtle Newsletter 78:2-7.

Eguchi, T., P. H. Dutton, S. A. Garner, and J. Alexander-Garner. 2006. Estimating juvenile survival rates and age at first nesting of leatherback turtles at St. Croix, U.S. Virgin Islands. Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Ehrhart, L. M. 1983. Marine turtles of the Indian River Lagoon system. Florida Scientist 46(3/4):337-346.

Ehrhart, L. M., W. E. Redfoot, and D. A. Bagley. 2007. Marine turtles of the central region of the Indian River Lagoon system, Florida. Florida Scientist 70(4):415-434.

Ehrhart, L. M., and R. G. Yoder. 1978. Marine turtles of Merritt Island National Wildlife Refuge, Kennedy Space Centre, Florida. Florida Marine Research Publications 33:25-30.

Epperly, S. P., J. Braun-McNeill, and P. M. Richards. 2007. Trends in catch rates of sea turtles in North Carolina, USA. Endangered Species Research 3:283-293.

Ferraroli, S., J.-Y. Georges, P. Gaspar, and Y. Le Maho. 2004. Where leatherback turtles meet fisheries. Nature 429:521-522.

FitzSimmons, N. N., L. W. Farrington, M. J. McCann, C. J. Limpus, and C. Moritz. 2003. Green turtle populations in the Indo-Pacific: A (genetic) view from microsatellites. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-536, Miami, FL.

Fleming, E. H. 2001. Swimming against the tide: Recent surveys of exploitation, trade, and management of marine turtles in the Northern Caribbean. TRAFFIC North America, Washington, D.C.

Foley, A. M., B. A. Schroeder, and S. L. MacPherson. 2008. Post-nesting migrations and resident areas of Florida loggerheads (*Caretta caretta*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-582, Miami, FL.

Foley, A. M., B. A. Schroeder, A. E. Redlow, K. J. Fick-Child, and W. G. Teas. 2005. Fibropapillomatosis in stranded green turtles (*Chelonia mydas*) from the eastern United

States (1980-98): Trends and associations with environmental factors. Journal of Wildlife Diseases 41(1):29-41.

Foley, A. M., K. E. Singel, P. H. Dutton, T. M. Summers, and A. E. Redlow. 2007. Characteristics of a green turtle (*Chelonia mydas*) assemblage in northwestern Florida determined during a hypothermic stunning event. Gulf of Mexico Science 25(2):13.

Formia, A. 1999. Les tortues marines de la Baie de Corisco. Canopée 14: i-ii.

Frazer, N. B., and L. M. Ehrhart. 1985. Preliminary growth models for green, (*Chelonia mydas*), and loggerhead, (*Caretta caretta*), turtles in the wild. Copeia 1985(1):73-79.

Fretey, J. 2001. Biogeography and conservation of marine turtles of the Atlantic coast of Africa. Convention on the Conservation of Migratory Species of Wild Animals (CMS), Secretariat, Bonn, Germany.

Fretey, J., A. Billes, and M. Tiwari. 2007. Leatherback, *Dermochelys coriacea*, nesting along the Atlantic coast of Africa. Chelonian Conservation and Biology 6(1):126-129.

Gaos, A. R., and coauthors. 2010. Signs of hope in the eastern Pacific: International collaboration reveals encouraging status for a severely depleted population of hawksbill turtle *Eretmochelys imbricata*. Oryx 44(4):595-601.

García-Muñoz, D., and L. Sarti. 2000. Reproductive cycles of leatherback turtles. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-436, Miami, FL.

Garduño-Andrade, M., V. Guzmán, E. Miranda, R. Briseño-Dueñas, and F. A. Abreu-Grobois. 1999. Increases in hawksbill turtle (*Eretmochelys imbricata*) nestings in the Yucatán Peninsula, Mexico, 1977-1996: Data in support of successful conservation? Chelonian Conservation and Biology 3(2):286-295.

Girard, C., A. D. Tucker, and B. Calmettes. 2009. Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: Dispersal in highly dynamic conditions. Marine Biology 156(9):1827-1839.

Glen, F., A. C. Broderick, B. J. Godley, and G. C. Hays. 2006. Thermal control of hatchling emergence patterns in marine turtles. Journal of Experimental Marine Biology and Ecology 334(1):31-42.

Glenn, L. 1996. The orientation and survival of loggerhead sea turtle hatchlings (*Caretta caretta* L.) in the nearshore environment. Unpublished Master's thesis. Florida Atlantic University, Boca Raton, FL.

Goff, G. P., and J. Lien. 1988. Atlantic leatherback turtles, *Dermochelys coriacea*, in cold water off Newfoundland and Labrador. Canadian Field-Naturalist 102:1-5.

Gonzalez Carman, V., and coauthors. 2011. Argentinian coastal waters: A temperate habitat for three species of threatened sea turtles. Marine Biology Research 7:500-508.

Graham, T. R. 2009. Scyphozoan jellies as prey for leatherback sea turtles off central California. Master's Thesis. San José State University, Moss Landing, CA.

Green, D. 1993. Growth rates of wild immature green turtles in the Galápagos Islands, Ecuador. Journal of Herpetology 27(3):338-341.

Greer Jr., A. E., J. D. Lazell Jr., and R. M. Wright. 1973. Anatomical evidence for a counter-current heat exchanger in the leatherback turtle (*Dermochelys coriacea*). Nature 244:181.

Groombridge, B., and R. Luxmoore. 1989. The green turtle and hawksbill (Reptilia: Cheloniidae): World status, exploitation and trade. Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, Lausanne, Switzerland.

Groombridge, B., and L. Wright (compilers). 1982. Kemp's ridley or Atlantic ridley, *Lepidochelys kempii* (Garman 1880). Pages 201-208 *in* The IUCN Amphibia-Reptilia Red Data Book. Part 1, Testudines, Crocodylia, Rhynchocephalia, volume 1. IUCN Conservation Monitoring Centre, Gland, Switzerland.

Guseman, J. L., and L. M. Ehrhart. 1991. Ecological geography of western Atlantic loggerheads and green turtles: Evidence from remote tag recoveries. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Technical Memorandum NMFS-SEFSC-302, Miami, FL.

Hart, K. M., M. M. Lamont, I. Fujisaki, A. D. Tucker, and R. R. Carthy. 2012. Common coastal foraging areas for loggerheads in the Gulf of Mexico: Opportunities for marine conservation. Biological Conservation 145(1):185-194.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology 13(5):923-932.

Hays, G. C., and coauthors. 2001. The diving behavior of green turtles undertaking oceanic migration to and from Ascension Island: Dive durations, dive profiles, and depth distribution. Journal of Experimental Biology 204:4093-4098.

Hays, G. C., and coauthors. 2002. Water temperature and internesting intervals for loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtles. Journal of Thermal Biology 27(5):429-432.

Hays, G. C., J. D. R. Houghton, and A. E. Myers. 2004. Pan-Atlantic leatherback turtle movements. Nature 429:522.

Heppell, S. S., and coauthors. 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chelonian Conservation and Biology 4(4):767-773.

Heppell, S. S., L. B. Crowder, and T. R. Menzel. 1999. Life table analysis of long-lived marine species with implications for conservation and management, volume 23. American Fisheries Society, Monterey, CA

Heppell, S. S., L. B. Crowder, and J. Priddy. 1995. Evaluation of a fisheries model for hawksbill sea turtle (*Eretmochelys imbricata*) harvest in Cuba. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-OPR-5, Raleigh, NC.

Heppell, S. S., M. L. Snover, and L. B. Crowder. 2003. Sea turtle population ecology. Pages 275-306 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, Vol. II. CRC Press, Boca Raton, FL.

Herbst, L. H. 1994. Fibropapillomatosis of marine turtles. Annual Review of Fish Diseases 4:389-425.

Herbst, L. H. 1995. An infectious etiology for green turtle fibropapillomatosis, volume 36. American Association for Cancer Research, Toronto, Ontario, Canada.

Hildebrand, H. H. 1963. Hallazgo del area de anidacion de la tortuga marina "lora", *Lepidochelys kempi* (Garman), en la costa occidental del Golfo de Mexico (Rept., Chel.). Ciencia, Mexico 22(4):105-112.

Hildebrand, H. H. 1982. A historical review of the status of sea turtle populations in the western Gulf of Mexico. Pages 447-453 *in* K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles: Proceedings of the World Conference on Sea Turtle Conservation. Smithsonian Institution Press, Washington, D.C.

Hillis, Z.-M., and A. L. Mackay. 1989. Research report on nesting and tagging of hawksbill sea turtles *Eretmocheys imbricata* at Buck Island Reef National Monument, U.S. Virgin Islands, 1987-88. U.S. Department of the Interior, National Park Service, Buck Island Reef National Monument, Division of Natural Resource Management, NPS PX 5380-8-0090, Christiansted, St. Croix, U.S. Virgin Islands.

Hilterman, M., E. Goverse, M. Godfrey, M. Girondot, and C. Sakimin. 2003. Seasonal sand temperature profiles of four major leatherback nesting beaches in the Guyana Shield. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Hirth, H. F. 1971. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus) 1758. Food and Agricultural Organization of the United Nations, FAO Fisheries Synopsis No. 85, Rome, Italy.

Hirth, H. F. 1997. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). University of Utah, Salt Lake City, UT.

Hirth, H. F., J. Kasu, and T. Mala. 1993. Observations on a leatherback turtle *Dermochelys coriacea* nesting population near Piguwa, Papua New Guinea. Biological Conservation 65(1):77-82.

Hirth, H. F., and E. M. A. Latif. 1980. A nesting colony of the hawksbill turtle (*Eretmochelys imbricata*) on Seil Ada Kebir Island, Suakin Archipelago, Sudan. Biological Conservation 17(2):125-130.

Houghton, J. D. R., A. C. Broderick, B. J. Godley, J. D. Metcalfe, and G. C. Hays. 2002. Diving behaviour during the internesting interval for loggerhead turtles *Caretta caretta* nesting in Cyprus. Marine Ecology Progress Series 227:63-70.

Houghton, J. D. R., T. K. Doyle, M. W. Wilson, J. Davenport, and G. C. Hays. 2006. Jellyfish aggregations and leatherback turtle foraging patterns in a temperate coastal environment. Ecology 87(8):1967-1972.

Hughes, G. R. 1996. Nesting of the leatherback turtle (*Dermochelys coriacea*) in Tongaland, KwaZulu-Natal, South Africa, 1963-1995. Chelonian Conservation and Biology 2(2):153-158.

Ingle, R. M., and F. G. W. Smith. 1974. Sea turtles: and the turtle industry of the West Indies, Florida and the Gulf of Mexico, Revised edition. University of Miami Press, Coral Gables, FL.

ISTS. 1995. Proceedings of the International Symposium on Sea Turtle Conservation Genetics. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-396, Miami, FL.

Jacobson, E. R. 1990. An update on green turtle fibropapilloma. Marine Turtle Newsletter 49:7-8.

Jacobson, E. R., and coauthors. 1989. Cutaneous fibropapillomas of green turtles (Chelonia mydas). Journal of Comparative Pathology 101(1):39-52.

Jacobson, E. R., S. B. Simpson Jr., and J. P. Sundberg. 1991. Fibropapillomas in green turtles. Pages 99-100 *in* G. H. Balazs, and S. G. Pooley, editors. Research Plan for Marine Turtle Fibropapilloma: Results of a December 1990 Workshop. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center, Honolulu, HI.

James, M. C., S. A. Eckert, and R. A. Myers. 2005. Migratory and reproductive movements of male leatherback turtles (*Dermochelys coriacea*). Marine Biology 147(4):845-853.

James, M. C., S. A. Sherrill-Mix, and R. A. Myers. 2007. Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. Marine Ecology Progress Series 337:245-254.

Johnson, S. A., and L. M. Ehrhart. 1994. Nest-site fidelity of the Florida green turtle. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-341, Miami, FL.

Johnson, S. A., and L. M. Ehrhart. 1996. Reproductive ecology of the Florida green turtle: Clutch frequency. Journal of Herpetology 30(3):407-410.

Jones, T. T., M. D. Hastings, B. L. Bostrom, D. Pauly, and D. R. Jones. 2011. Growth of captive leatherback turtles, *Dermochelys coriacea*, with inferences on growth in the wild: Implications for population decline and recovery. Journal of Experimental Marine Biology and Ecology 399(1):84-92.

Keinath, J. A., and J. A. Musick. 1993. Movements and diving behavior of a leatherback turtle, *Dermochelys coriacea*. Copeia 1993(4):1010-1017.

Lagueux, C. J. 2001. Status and distribution of the green turtle, *Chelonia mydas*, in the wider Caribbean region. Pages 32-35 *in* K. L. Eckert, and F. A. Abreu-Grobois, editors. Proceedings of the Regional Meeting: "Marine Turtle Conservation in the Wider Caribbean Region: A Dialogue for Effective Regional Management". WIDECAST, IUCN-MTSG, WWF, UNEP-CEP, St. Croix, U.S. Virgin Islands.

Laurent, L., and coauthors. 1998. Molecular resolution of marine turtle stock composition in fishery by-catch: A case study in the Mediterranean. Molecular Ecology 7:1529-1542.

Law, R. J., and coauthors. 1991. Concentrations of trace metals in the livers of marine mammals (seals, porpoises and dolphins) from waters around the British Isles. Marine Pollution Bulletin 22(4):183-191.

León, Y. M., and C. E. Diez. 1999. Population structure of hawksbill turtles on a foraging ground in the Dominican Republic. Chelonian Conservation and Biology 3(2):230-236.

León, Y. M., and C. E. Díez. 2000. Ecology and population biology of hawksbill turtles at a Caribbean feeding ground. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NMFS-SEFSC-436, Miami, FL.

Lezama, C. 2009. Impacto de la pesquería artesanal sobre la tortuga verde (*Chelonia mydas*) en las costas del Río de la Plata exterior. Universidad de la República, Montevideo, Uruguay.

Lima, E. H. S. M., M. T. D. Melo, and P. C. R. Barata. 2010. Incidental capture of sea turtles by the lobster fishery off the Ceará Coast, Brazil. Marine Turtle Newsletter 128:16-19.

Limpus, C. J. 1992. The hawksbill turtle, *Eretmochelys imbricata*, in Queensland: Population struture within a southern Great Barrier Reef feeding ground. Wildlife Research 19(4):489-505.

Limpus, C. J., and J. D. Miller. 2000. Final report for Australian hawksbill turtle population dynamics project. Queensland Parks and Wildlife Service, Queensland, Australia.

López-Barrera, E. A., G. O. Longo, and E. L. A. Monteiro-Filho. 2012. Incidental capture of green turtle (*Chelonia mydas*) in gillnets of small-scale fisheries in the Paranaguá Bay, Southern Brazil. Ocean and Coastal Management 60:11-18.

López-Mendilaharsu, M., and coauthors. 2006. Biología, ecología y etología de las tortugas marinas en la zona costera uruguaya. Bases para la conservación y el manejo de la costa uruguaya:11.

Lund, P. F. 1985. Hawksbill turtle (*Eretmochelys imbricata*) nesting on the east coast of Florida. Journal of Herpetology 19(1):164-166.

Lutcavage, M. E., P. Plotkin, B. Witherington, and P. L. Lutz. 1997. Human impacts of sea turtle survival. Pages 387-404 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Mackay, A. L. 2006. 2005 Sea turtle monitoring program the East End beaches (Jack's, Isaac's, and East End Bay) St. Croix, U.S. Virgin Islands: Final report. West Indies Marine Animal Research and Conservation Service, Inc, Christiansted, St. Croix, U.S. Virgin Islands.

MAFMC and ASMFC. 1998. Amendment 1 to the Bluefish Fishery Management Plan (includes Environmental Impact Statement and Regulatory Impact Review), Vol 1. Mid-Atlantic Fishery Management Council and the Atlantic States Marine Fisheries Commission.

Maharaj, A. M. 2004. A comparative study of the nesting ecology of the leatherback turtle *Dermochelys coriacea* in Florida and Trinidad. Master's Thesis. University of Central Florida, Orlando, FL.

Marcovaldi, N., B. B. Gifforni, H. Becker, F. N. Fiedler, and G. Sales. 2009. Sea turtle interactions in coastal net fisheries in Brazil. Western Pacific Regional Fishery Management Council, IUCN, Southeast Asian Fisheries Development Center, Indian Ocean - South-East Asian Marine Turtle MoU, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Honolulu, HI.

Márquez M., R. 1990. Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date. Food and Agricultural Organization of the United Nations, FAO Fisheries Synopsis No. 125, Rome, Italy.

Márquez M., R. 1994. Synopsis of biological data on the Kemp's ridley sea turtle, *Lepidochelys kempii* (Garman, 1880). Instituto Nacional de la Pesca, U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region and U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-343, Miami, FL.

Martínez, L. S., and coauthors. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology 6(1):70-78.

Matos, R. 1986. Sea turtle hatchery project with specific reference to the leatherback turtle (*Dermochelys coriacea*), Humacao, Puerto Rico 1986. Puerto Rico Department of Natural Resources, de Tierra, Puerto Rico.

Mayor, P. A., B. Phillips, and Z.-M. Hillis-Starr. 1998. Results of the stomach content analysis on the juvenile hawksbill turtles of Buck Island Reef National Monument, U.S.V.I. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-415, Miami, FL.

McDonald, D. L., and P. H. Dutton. 1996. Use of PIT tags and photoidentification to revise remigration estimates of leatherback turtles (*Dermochelys coriacea*) nesting in St. Croix, U.S. Virgin Islands, 1979-1995. Chelonian Conservation and Biology 2(2):148-152.

Mckenzie, C., B. J. Godley, R. W. Furness, and D. E. Wells. 1999. Concentrations and patterns of organochlorine contaminants in marine turtles from Mediterranean and Atlantic waters. Marine Environmental Research 47:117-135.

McMichael, E., R. R. Carthy, and J. A. Seminoff. 2003. Evidence of homing behavior in juvenile green turtles in the northeastern Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Meylan, A. B. 1988. Spongivory in hawksbill turtles: A diet of glass. Science 239(4838):393-395.

Meylan, A. B. 1999a. International movements of immature and adult hawksbill turtles (*Eretmochelys imbricata*) in the Caribbean region. Chelonian Conservation and Biology 3(2):189-194.

Meylan, A. B. 1999b. Status of the hawksbill turtle (*Eretmochelys imbricata*) in the Caribbean region. Chelonian Conservation and Biology 3(2):177-184.

Meylan, A. B., and M. Donnelly. 1999. Status justification for listing the hawksbill turtle (*Eretmochelys imbricata*) as critically endangered on the 1996 IUCN Red List of Threatened Animals. Chelonian Conservation and Biology 3(2):200-204.

Meylan, A. B., B. Schroeder, and A. Mosier. 1994. Marine turtle nesting activity in the state of Florida, 1979-1992. U.S. Department of Commerce, National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-351, Miami, FL.

Meylan, A. B., B. A. Schroeder, A. Mosier, and I. Florida Marine Research. 1995. Sea turtle nesting activity in the state of Florida, 1979-1992. State of Florida, Department. of Environmental Protection, Florida Marine Research Institute, 0095-0157, Saint Petersburg, FL.

Meylan, A. B., B. E. Witherington, B. Brost, R. Rivera, and P. S. Kubilis. 2006. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of *Caretta*, *Chelonia*, and *Dermochelys*, Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation edition. International Sea Turtle Society, Athens, Greece.

Miller, J. D. 1997. Reproduction in sea turtles. Pages 51-81 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Milliken, T., and H. Tokunaga. 1987. The Japanese sea turtle trade 1970-1986. TRAFFIC (Japan), IUCN-Rep-1987-081, Washington, D.C.

Milton, S. L., and P. L. Lutz. 2003. Physiological and genetic responses to environmental stress. Pages 163-197 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, Vol. II. CRC Press, Boca Raton, FL.

Mo, C. L. 1988. Effect of bacterial and fungal infection on hatching success of olive ridley sea turtle eggs. Report to the U.S. World Wildlife Fund.

Moncada-Gavilán, F. 2001. Status and distribution of the loggerhead turtle, *Caretta caretta*, in the wider Caribbean region. Pages 36-40 *in* K. L. Eckert, and F. A. Abreu-Grobois, editors. Proceedings of the Regional Meeting: "Marine Turtle Conservation in the Wider Caribbean Region - A Dialogue for Effective Regional Management". WIDECAST, IUCN-MTSG, WWF, UNEP-CEP, St. Croix, U.S. Virgin Islands.

Moncada, F., E. Carrillo, A. Saenz, and G. Nodarse. 1999. Reproduction and nesting of the hawksbill turtle, *Eretmochelys imbricata*, in the Cuban Archipelago. Chelonian Conservation and Biology 3(2):257-263.

Moncada, F. B., and coauthors. 2010. Movement patterns of loggerhead turtles *Caretta caretta* in Cuban waters inferred from flipper tag recaptures. Endangered Species Research 11(1):61-68.

Monzón-Argüello, C., and coauthors. 2010. Evidence from genetic and Lagrangian drifter data for transatlantic transport of small juvenile green turtles. Journal of Biogeography 37(9):1752-1766.

Mortimer, J. A., and A. Carr. 1987. Reproduction and migrations of the Ascension Island green turtle (*Chelonia mydas*). Copeia 1987(1):103-113.

Mortimer, J. A., and coauthors. 2003. Growth rates of immature hawksbills (*Eretmochelys imbricata*) at Aldabra Atoll, Seychelles (Western Indian Ocean). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Mortimer, J. A., and M. Donnelly. 2008. Hawksbill turtle (*Eretmochelys imbricata*). IUCN Red List status assessment Marine Turtle Specialist Group, International Union for Conservation of Nature and Natural Resources.

Mrosovsky, N., G. D. Ryan, and M. C. James. 2009. Leatherback turtles: The menace of plastic. Marine Pollution Bulletin 58(2):287-289.

Murphy, T. M., and S. R. Hopkins. 1984. Aerial and ground surveys of marine turtle nesting beaches in the southeast region, U.S.: Final report to the National Marine Fisheries Service. LaMER, NMFS Contract Number NA83-GA-C-00021, Green Pond, SC.

Musick, J. A., and C. J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pages 137-163 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Naro-Maciel, E., J. H. Becker, E. H. S. M. Lima, M. Â. Marcovaldi, and R. DeSalle. 2007. Testing dispersal hypotheses in foraging green sea turtles (*Chelonia mydas*) of Brazil. Journal of Heredity 98(1):29-39.

Naro-Maciel, E., and coauthors. 2012. The interplay of homing and dispersal in green turtles: A focus on the southwestern Atlantic. Journal of Heredity 103(6):792-805.

National Research Council. 1990a. Decline of the Sea Turtles: Causes and Prevention. National Academy Press, Washington, D.C.

National Research Council. 1990b. Sea turtle mortality associated with human activities. Pages 74-117 *in* National Research Council Committee on Sea Turtle Conservation, editor. Decline of the Sea Turtles: Causes and Prevention. National Academy Press, Washington, D.C.

NMFS. 1995. Biological Opinion on United States Coast Guard vessel and aircraft activities along the Atlantic coast. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

NMFS. 1996. Reinitiation - Biological Opinion on United States Coast Guard vessel and aircraft activities along the Atlantic coast. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

NMFS. 1997a. Biological Opinion on Navy activities off the southeastern United States along the Atlantic coast. U.S. Department of Commerce, National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 1997b. Biological Opinion on the continued hopper dredging of channels and borrow areas in the southeastern United States. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 2001. Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the western North Atlantic. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-455, Miami, FL.

NMFS. 2009. An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, PRD-08/09-14, Miami, FL.

NMFS. 2011a. Final recovery plan for the sei whale (*Balaenoptera borealis*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS. 2011b. Preliminary summer 2010 regional abundance estimate of loggerhead turtles (*Caretta caretta*) in northwestern Atlantic Ocean continental shelf waters. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Reference Document 11-03, Woods Hole, MA.

NMFS. 2012a. Reinitiation - Biological Opinion on the continued implementation of the sea turtle conservation regulations, as proposed to be amended, and the continued authorization of the southeast U.S. shrimp fisheries in federal waters under the Magnuson-Stevens Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 2012b. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments - 2011. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, NOAA Technical Memorandum NMFS-NE-221, Woods Hole, MA.

NMFS. 2013. Biological Opinion for the U.S. Navy's Atlantic Fleet training and testing activities from November 2013 through November 2018; and the National Marine Fisheries Services' promulgation of regulations and issuance of letters of authorization pursuant to the Marine Mammal Protection Act for the U.S. Navy to "take" marine mammals incidental to Atlantic Fleet training and testing activities from November 2013 through November 2018. U.S. Department of Commerce, National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS. 2015. Reinititation - Biological Opinion on the continued authorization of the fishery management plan for coastal migratory pelagic resources in the Atlantic and Gulf of Mexico under the Magnuson-Stevens Fishery Management and Conservation Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2015-15985, Saint Petersburg, FL.

NMFS. 2018. Biological Opinion on Bogue Banks Master Beach Nourishment Plan. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS, USFWS, and SEMARNAT. 2011. Bi-national recovery plan for the Kemp's ridley sea turtle (*Lepidochelys kempii*), Second revision. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 1991. Recovery plan for U.S. population of Atlantic green turtle (*Chelonia mydas*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Washington, D.C.

NMFS and USFWS. 1992. Recovery plan for the leatherback turtles *Dermochelys coriacea* in the U.S. Caribbean, Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Washington, D.C.

NMFS and USFWS. 1993. Recovery plan for the hawksbill turtle *Eretmochelys imbricata* in the U.S. Caribbean, Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Saint Petersburg, FL.

NMFS and USFWS. 1995. Status reviews for sea turtles listed under the Endangered Species Act of 1973. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 1998. Recovery plan for U.S. Pacific populations of the hawksbill turtle (*Eretmochelys imbricata*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2007a. Green sea turtle (*Chelonia mydas*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2007b. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2008. Recovery plan for the Northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*), Second revision. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2013a. Hawksbill sea turtle (*Eremochelys imbricata*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2013b. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fiseries Service, Silver Spring, MD.

NMFS and USFWS. 2015. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

Ogren, L. H. 1989a. Distribution of juvenile and subadult Kemp's ridley sea turtles: Preliminary results from 1984-1987 surveys. Texas A&M University Sea Grant College Program, TAMU-SG-89-105, Galveston, TX.

Ogren, L. H. 1989b. Status report of the green turtle. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center, NOAA Technical Memorandum NMFS-SEFC-226, Panama City, FL.

Paladino, F. V., M. P. O'Connor, and J. R. Spotila. 1990. Metabolism of leatherback turtles, gigantothermy, and thermoregulation of dinosaurs. Nature 344:858-860.

Parsons, J. J. 1972. The hawksbill turtle and the tortoise shell trade. Pages 45-60 *in* Études de Géographie Tropicale Offertes a Pierre Gourou. Walter de Gruyter GmbH & Co, Paris, France.

Pike, D. A., R. L. Antworth, and J. C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the loggerhead sea turtle, *Caretta caretta*. Journal of Herpetology 40(1):91-94.

Piniak, W. D., K. L. Eckert, M. Palmer, and P. Kramer. 2007. An atlas of sea turtle nesting habitat for the wider Caribbean region. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy, WIDECAST Technical Report No. 6, Beaufort, NC.

Plotkin, P., and A. F. Amos. 1990. Effects of anthropogenic debris on sea turtles in the northwestern Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOAA Technical Memorandum NMFS SWFSC-154, Honolulu, HI.

Plotkin, P. T. 2003. Adult migrations and habitat use. Pages 225-241 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, Vol. II. CRC Press, Boca Raton, FL.

Plotkin, P. T., and A. F. Amos. 1988. Entanglement in and ingestion of marine debris by sea turtles stranded along the south Texas coast. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center, NOAA Technical Memorandum NMFS-SEFC-214, Miami, FL.

Pritchard, P. C. H. 1969. The survival status of ridley sea-turtles in America. Biological Conservation 2(1):13-17.

Pritchard, P. C. H., and coauthors. 1983. Manual of sea turtle research and conservation techniques, Second edition. Center for Environmental Education, Washington, D.C.

Pritchard, P. C. H., and P. Trebbau. 1984. The Turtles of Venezuela, Contributions to herpetology edition. Society for the Study of Amphibians and Reptiles.

Prosdocimi, L., V. González Carman, D. A. Albareda, and M. I. Remis. 2012. Genetic composition of green turtle feeding grounds in coastal waters of Argentina based on mitochondrial DNA. Journal of Experimental Marine Biology and Ecology 412:37-45.

Rabalais, N. N., and coauthors. 2002. Nutrient-enhanced productivity in the northern Gulf of Mexico: Past, present and future. Hydrobiologia 475(1):39-63.

Rhodin, A. G. J. 1985. Comparative chondro-osseous development and growth in marine turtles. Copeia 1985(3):752-771.

Richards, P. M., and coauthors. 2011. Sea turtle population estimates incorporating uncertainty: A new approach applied to western North Atlantic loggerheads *Caretta caretta*. Endangered Species Research 15(2):151-158.

Richardson, J. I., R. Bell, and T. H. Richardson. 1999. Population ecology and demographic implications drawn from an 11-year study of nesting hawksbill turtles, *Eretmochelys imbricata*, at Jumby Bay, Long Island, Antigua, West Indies. Chelonian Conservation and Biology 3(2):244-250.

Rivalan, P., and coauthors. 2005. Trade-off between current reproductive effort and delay to next reproduction in the leatherback sea turtle. Oecologia 145(4):564-574.

Ruben, H. J., and S. J. Morreale. 1999. Draft biological assessment for sea turtles in New York and New Jersey Harbor Complex. Unpublished biological assessment submitted to National Marine Fisheries Service. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

Sakamoto, W., and coauthors. 1990. Deep diving behavior of the loggerhead turtle near the frontal zone. Nippon Suisan Gakkaishi 56(9):1435-1443.

Salmon, M., and J. Wyneken. 1987. Orientation and swimming behavior of hatchling loggerhead turtles *Caretta caretta* L. during their offshore migration. Journal of Experimental Marine Biology and Ecology 109(2):137-153.

Santidrián Tomillo, P., and coauthors. 2007. Reassessment of the leatherback turtle (*Dermochelys coriacea*) nesting population at Parque Nacional Marino Las Baulas, Costa Rica: Effects of conservation efforts. Chelonian Conservation and Biology 6(1):54-62.

Schmid, J. R., and W. J. Barichivich. 2006. *Lepidochelys kempii* – Kemp's ridley. Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3:128-141.

Schmid, J. R., and A. Woodhead. 2000. Von Bertalanffy growth models for wild Kemp's ridley turtles: Analysis of the NMFS Miami Laboratory tagging database. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-444, Miami, FL.

Schroeder, B. A., and A. M. Foley. 1995. Population studies of marine turtles in Florida Bay. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-361, Miami, FL.

Schulz, J. P. 1975. Sea turtles nesting in Surinam. Zoologische Verhandelingen 143:141.

Schulz, J. P. 1982. Status of sea turtle populations nesting in Surinam with notes on sea turtles nesting in Guyana and French Guiana. Pages 435-437 *in* K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles: Proceedings of the World Conference on Sea Turtle Conservation. Smithsonian Institution Press, Washington, D.C.

Seminoff, J. A., and coauthors. 2015. Status review of the green turtle (*Chelonia mydas*) under the Endangered Species Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOAA Technical Memorandum NMFS-SWFSC-539, La Jolla, CA.

Shaver, D. J. 1994. Relative abundance, temporal patterns, and growth of sea turtles at the Mansfield Channel, Texas. Journal of Herpetology 28(4):491-497.

Shenker, J. M. 1984. Scyphomedusae in surface waters near the Oregon coast, May-August, 1981. Estuarine, Coastal and Shelf Science 19(6):619-632.

Shillinger, G. L., and coauthors. 2008. Persistent leatherback turtle migrations present opportunities for conservation. PLoS Biology 6(7):1408-1416.

Shoop, C. R., and R. D. Kenney. 1992. Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. Herpetological Monographs 6:43-67.

Snover, M. L. 2002. Growth and ontogeny of sea turtles using skeletochronology: Methods, validation and application to conservation. Doctor of Philosophy. Duke University, Durham, NC.

Southwood, A. L., R. D. Andrews, F. V. Paladino, and D. R. Jones. 2005. Effects of diving and swimming behavior on body temperatures of Pacific leatherback turtles in tropical seas. Physiological and Biochemical Zoology 78(2):285-297.

Spotila, J. R. 2004. Sea Turtles: A Complete Guide to their Biology, Behavior, and Conservation. Johns Hopkins University Press, Baltimore, MD.

Spotila, J. R., and coauthors. 1996. Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? Chelonian Conservation and Biology 2(2):209-222.

Spotila, J. R., R. D. Reina, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. 2000. Pacific leatherback turtles face extinction. Nature 405:529-530.

Stapleton, S., and C. Stapleton. 2006. Tagging and nesting research on hawksbill turtles (*Eretmochelys imbricata*) at Jumby Bay, Long Island, Antigua, West Indies: 2005 Annual report. Jumby Bay Hawksbill Project, Wider Caribbean Sea Turtle Conservation Network.

Starbird, C. H., A. Baldridge, and J. T. Harvey. 1993. Seasonal occurrence of leatherback sea turtles (*Dermochelys coriacea*) in the Monterey Bay region, with notes on other sea turtles, 1986-1991. California Fish and Game 79(2):54-62.

Starbird, C. H., and M. M. Suarez. 1994. Leatherback sea turtle nesting on the north Vogelkop coast of Irian Jaya and the discovery of a leatherback sea turtle fishery on Kei Kecil Island. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-351, Miami, FL.

Stewart, K., and C. Johnson. 2006. *Dermochelys coriacea* - Leatherback sea turtle. Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3:144-157.

Stewart, K., C. Johnson, and M. H. Godfrey. 2007. The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. Herpetological Journal 17(2):123-128.

Stewart, K. R., and J. Wyneken. 2004. Predation risk to loggerhead hatchlings at a high-density nesting beach in southeast Florida. Bulletin of Marine Science 74(2):325-335.

Steyermark, A. C., and coauthors. 1996. Nesting leatherback turtles at Las Baulas National Park, Costa Rica. Chelonian Conservation and Biology 2(2):173-183.

Storelli, M. M., G. Barone, A. Storelli, and G. O. Marcotrigiano. 2008. Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. Chemosphere 70(5):908-913.

Storelli, M. M., E. Ceci, and G. O. Marcotrigiano. 1998. Distribution of heavy metal residues in some tissues of *Caretta caretta* (Linnaeus) specimen beached along the Adriatic Sea (Italy). Bulletin of Environmental Contamination and Toxiocology 60:546-552.

Suchman, C. L., and R. D. Brodeur. 2005. Abundance and distribution of large medusae in surface waters of the northern California Current. Deep Sea Research Part II: Topical Studies in Oceanography 52(1–2):51-72.

Tiwari, M., B. P. Wallace, and M. Girondot. 2013. *Dermochelys coriacea* (Northwest Atlantic Ocean subpopulation), Leatherback. The IUCN Red List of Threatened Species, International Union for Conservation of Nature and Natural Resources.

Troëng, S. 1998. Poaching threatens the green turtle rookery at Tortuguero, Costa Rica. Marine Turtle Newsletter 79:11-12.

Troëng, S., D. Chacón, and B. Dick. 2004. Possible decline in leatherback turtle *Dermochelys coriacea* nesting along the coast of Caribbean Central America. Oryx 38(4):395-403.

Troëng, S., D. Chacón, and B. Dick. 2005. Leatherback turtle *Dermochelys coriacea* nesting along the Caribbean coast of Costa Rica. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-528, Miami, FL.

Troëng, S., E. Harrison, D. Evans, A. de Haro, and E. Vargas. 2007. Leatherback turtle nesting trends and threats at Tortuguero, Costa Rica. Chelonian Conservation and Biology 6(1):117-122.

Troëng, S., and E. Rankin. 2005. Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biological Conservation 121:111-116.

Tucker, A. D. 1988. A summary of leatherback turtle *Dermochelys coriacea* nesting at Culebra, Puerto Rico from 1984-1987 with management recommendations. U.S. Department of the Interior, Fish and Wildlife Service, Athens GA.

Tucker, A. D. 2010. Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: Implications for stock estimation. Journal of Experimental Marine Biology and Ecology 383(1):48-55.

Turtle Expert Working Group. 1998. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the western North Atlantic. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-409, Miami, FL.

Turtle Expert Working Group. 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-444, Miami, FL.

Turtle Expert Working Group. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-555, Miami, FL.

Turtle Expert Working Group. 2009. An assessment of the loggerhead turtle population in the western North Atlantic ocean. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-575, Miami, FL.

van Dam, R., and C. E. Díez. 1997. Predation by hawksbill turtles on sponges at Mona Island, Puerto Rico, volume 2. Smithsonian Tropical Research Institute, Balboa, Panama.

van Dam, R., L. S. M., and D. J. Pares. 1992. The hawksbills of Mona Island, Puerto Rico: Report for 1990. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-302, Miami, FL.

van Dam, R., L. S. M., and B. P. R. 1990. Sea turtle biology and conservation on Mona Island, Puerto Rico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFC-278, Miami, FL.

van Dam, R. P., and C. E. Diez. 1998. Home range of immature hawksbill turtles (*Eretmochelys imbricata* (Linnaeus)) at two Caribbean islands. Journal of Experimental Marine Biology and Ecology 220:15-24.

Vargo, S. L., P. L. Lutz, D. K. Odell, E. S. van Vleet, and G. D. Bossart. 1986. Final report: Study of the effects of oil on marine turtles. Florida Institute of Oceanography, MMS Contract Number 14-12-0001-30063, Saint Petersburg, FL.

Weijerman, M., L. H. G. van Tienen, A. D. Schouten, and W. E. J. Hoekert. 1996. Sea turtles of Galibi, Suriname. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-412, Miami, FL.

Weishampel, J. F., D. A. Bagley, and L. M. Ehrhart. 2004. Earlier nesting by loggerhead sea turtles following sea surface warming. Global Change Biology 10:1424-1427.

Weishampel, J. F., D. A. Bagley, L. M. Ehrhart, and B. L. Rodenbeck. 2003. Spatiotemporal patterns of annual sea turtle nesting behaviors along an East Central Florida beach. Biological Conservation 110(2):295-303.

Wenzel, F. W., D. K. Mattila, and P. J. Clapham. 1988. *Balaenoptera musculus* in the Gulf of Maine. Marine Mammal Science 4(2):172-175.

Wershoven, J. L., and R. W. Wershoven. 1992. Juvenile green turtles in their nearshore habitat of Broward County, Florida: A five year review. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-302, Miami, FL.

Whiting, S. D. 2000. The foraging ecology of juvenile green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) sea turtles in north-western Australia. Doctor of Philosophy. Northern Territory University, Darwin, Australia.

Wilkinson, C. R. 2004. Status of Coral Reefs of the World: 2004, volume 2. Australian Institute of Marine Science, Townsville, Australia.

Witherington, B. E. 1991. Orientation of hatchling loggerhead turtles at sea off artificially lighted and dark beaches. Journal of Experimental Marine Biology and Ecology 149(1):1-11.

Witherington, B. E. 2002. Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. Marine Biology 140(4):843-853.

Witherington, B. E., M. Bresette, and R. M. Herren. 2006. *Chelonia mydas* - Green turtle. Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3 3:90-104.

Witherington, B. E., and L. M. Ehrhart. 1989a. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon system, Florida. Copeia (3):696-703.

Witherington, B. E., and L. M. Ehrhart. 1989b. Status and reproductive characteristics of green turtles (*Chelonia mydas*) nesting in Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center, NOAA Technical Memorandum NMFS-SEFSC-226, Panama City, FL.

Witt, M. J., and coauthors. 2007. Prey landscapes help identify foraging habitats for leatherback turtles in the NE Atlantic. Marine Ecology Progress Series (337):231-243.

Witt, M. J., B. J. Godley, A. C. Broderick, R. Penrose, and C. S. Martin. 2006. Leatherback turtles, jellyfish and climate change in the northwest Atlantic: Current situation and possible future scenarios. Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Witzell, W. N. 1983. Synopsis of biological data on the hawksbill sea turtle, *Eretmochelys imbricata* (Linnaeus, 1766). Food and Agricultural Organization of the United Nations, FAO Fisheries Synopsis No. 137, Rome, Italy.

Witzell, W. N. 2002. Immature Atlantic loggerhead turtles (*Caretta caretta*): Suggested changes to the life history model. Herpetological Review 33(4):266-269.

Wyneken, J., L. Fisher, M. Salmon, and S. Weege. 2000. Managing relocated sea turtle nests in open-beach hatcheries. Lessons in hatchery design and implementation in Hillsboro Beach, Broward County, Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheres Service, NOAA Technical Memorandum NMFS-SEFSC-443, Miami, FL.

Wyneken, J., and M. Salmon. 1992. Frenzy and post-frenzy swimming activity in loggerhead, leatherback, and green sea turtles. Copeia (2):478-484.

Zinno, F. R. 2012. Captura incidental de tortugas marinas en Bajos del Solís, Uruguay. Profundización en Ecología. Universidad de la República Uruguay, Montevideo, Uruguay.

Zug, G. R., and R. E. Glor. 1998. Estimates of age and growth in a population of green sea turtles (*Chelonia mydas*) from the Indian River Lagoon system, Florida: A skeletochronological analysis. Canadian Journal of Zoology 76(8):1497-1506.

Zug, G. R., and J. F. Parham. 1996. Age and growth in leatherback turtles, *Dermochelys coriacea*: A skeletochronological analysis. Chelonian Conservation and Biology 2(2):244-249.

Zurita, J. C., and coauthors. 2002. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Zwinenberg, A. J. 1977. Kemp's ridley, *Lepidochelys kempii* (Garman, 1880), undoubtedly the most endangered marine turtle today (with notes on the current status of *Lepidochelys olivacea*). Bulletin of the Maryland Herpetological Society 13(3):170-192.

APPENDIX A: PROJECT DESIGN CRITERIA

A. Material

1. Materials used for artificial reef building shall follow the NCDMF Guidelines and Specifications for Acceptable Reef Materials, the 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 the NOAA/NMFS National Artificial Reef Plan, and NMFS SERO artificial reef guidance.
2. 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 in length. 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 in length if the third side is lowered to allow a 4 ft length opening edge on that third side (see Figure 1). For instance, this would require a pyramid module with a 10 ft base that is 8 ft in height to 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. 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.
3. Vessels may also be utilized as artificial reef building material. These may be composed of ferro-cement or steel, though steel vessels are the most commonly used and preferred. Derelict vessels and military surplus shall have openings created on all exposed sides adequate to prevent entrapment of listed species.
4. The use of tires, Fish Aggregating Devices (FADs), post-use sanitary sewer materials, automobiles and other civilian vehicles, white goods (refrigerators, washers, etc.) boat molds, floatables, loose organic material and general demolition debris, other than clean concrete units to form reefs, are not authorized.
5. Derelict automobiles shall not be used.
6. Materials should be of sufficient size and relative density to not move from the reef site post-deployment.
7. All materials used for construction of reefs will be clean and free of petroleum and other hydrocarbons (oil, grease, asphalt and creosote), toxic residues (mercury, cadmium and lead) and loose, free floating material and other deleterious substances and/or in compliance with criteria established by the U.S. EPA.
8. All concrete must be fully cured to ensure environmental compatibility.

9. For secondary-use, recycled concrete and similar materials other than bridge spans, all steel reinforcement rods (rebar) must be cut at the base of the concrete so that no metal protrudes from the concrete's surface. Steel reinforcement rods will be cut from bridge spans at the base of the concrete to the extent possible and will not protrude more than 3 inches.
10. Pursuant to the U.S. Environmental Protection Agency (EPA) Best Management Practices, 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.

B. Deployment

1. Deployments will be conducted during daylight hours when lighting, weather, and sea conditions allow for visual monitoring of the project area.
2. The permittee shall not deploy artificial reef materials until an 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.
3. The Permittee shall follow the NMFS Sea Turtle and Smalltooth Sawfish Construction Conditions, Dated March 23, 2006, and will apply these measures to all ESA-listed species.
4. Deployment activities will not commence until the project supervisor reports that no sea turtles, marine mammals (North Atlantic right whales have additional restrictions listed below), or other ESA-listed species have been sighted within 300 ft of the active deployment site (i.e., barge carrying material or moored vessel to be scuttled) for at least 20 minutes. Deployment activities will cease immediately if sea turtles, marine mammals, or other ESA-listed species are sighted within 300 ft 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. NCDMF is required to participate in the Right Whale Early Warning System to protect North Atlantic right whales. If a right whale or any other species of whale is reported within the area, then the contractor will be required to follow the NMFS's Southeast Region Vessel Strike Avoidance Measures and Reporting for Mariners except where specific measures below are in conflict, in which case the measures in this Opinion govern. To review these requirements go to: https://www.fisheries.noaa.gov/southeast/consultations/regulations-policies-and-guidance. By law, vessels shall maintain a 500-yd buffer between the vessel and any North Atlantic right whale [as required by federal regulation 50 CFR 224.103 (c)].
5. NCDMF will follow an in-water construction moratorium on AR-130, AR-140, and AR-145 (and all newly constructed ocean reefs that may have overwintering sturgeon) from January 1 to March 31 to protect overwintering sturgeon. They will also follow a September 30th – February 1st in-water work moratorium during active sturgeon migration periods for AR-291, AR-392, and AR-396.

6. Reef structures shall be sited and installed in accordance with the permitted boundaries and site clearances.
7. The placement of material within any jurisdictional wetland, submerged aquatic vegetation bed, coral reefs, oyster reefs, mussel beds, scallop beds, clam beds or live bottoms (areas supporting the growth of sponges, sea fans, soft coral and other sessile macro invertebrates generally associated with rock outcrops) is not authorized.
8. Reef structures shall not be placed on natural hard-bottom habitat, and the permittee shall maintain a deployment buffer of at least 200 ft from any submerged aquatic resources, including seagrasses, macroalgae, hard or soft coral (including coral reefs), sponges, oysters, or hard bottom 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.
9. All material placed to construct reefs (specifically designed reef modules) will be selected and placed so as to avoid the movement of reef materials due to sea conditions or currents. The Permittee will be responsible for any materials which are moved by sea conditions or which break loose from reefs, and the Permittee will be responsible for any damage caused by such materials. No individual artificial reef component (i.e., prefabricated module, concrete piece, etc.) will weigh less than 500 pounds, 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).
10. The deployment of artificial reef material incorporating any mid-water (i.e., any distance above artificial reef structure) fish-aggregating device, excluding a maintained site navigational buoy, is not authorized under this consultation.
11. The use of explosives to deploy materials is not authorized.

C. Vessel Movement

1. Speed for all vessels 33 ft and greater involved in placing the reef material is 10 knots or less all year round.
2. If a protected species is seen within 300 ft of the vessel, all appropriate precautions shall be implemented to avoid a collision (for North Atlantic right whale regulations require maintaining a distance of 1500 ft). These precautions shall include cessation of any vessel movement when closer than 50 ft of a protected species (excluding at times when movement is required for safe navigation [e.g., transiting inlets]). Operation may not resume until the protected species has departed the immediate area of its own volition.

D. Critical Habitat

In addition to the PDCs listed above, the following PDCs apply to loggerhead sea turtle (NWA DPS), North Atlantic right whale, and Atlantic sturgeon Critical Habitat:

1. Loggerhead Sea Turtle (NWA DPS) Critical Habitat

i. No artificial reef material will be deployed in any nearshore area 1,100 ft of any identified sea turtle nesting beach that predominantly consists of sandy benthic habitat.
ii. No emergent artificial reef material will be authorized in identified loggerhead sea turtle nearshore reproductive critical habitat areas. Any artificial reef material deployed within these critical habitat areas and within 1,100 ft of the beach at mean low water (MLW) must provide at least 4 ft of surface clearance at MLW, a maximum reef section length of 50 ft, and the project must include gaps free of any material at a 1:1 ratio (e.g., for every 25 ft of contiguous artificial reef material, a 25-ft gap clear of any material must be created).

2. Atlantic Sturgeon Critical Habitat

i. No new reefs will be constructed in Atlantic sturgeon designated critical habitat.

3. North Atlantic Right Whale Critical Habitat

i. No artificial reefs can be placed in water shallower than 30 ft deep
ii. The maximum footprint of new reefs shall be 1 nmi^2. If a new reef is added to an existing artificial reef, the total footprint of the combined reefs must not exceed 1 nmi^2.
iii. Density of newly permitted reefs shall not exceed 2 reefs (old or new) per 10 nmi^2.
iv. All effort should be made to avoid placing reef material during North Atlantic right whale calving season (November 15 through April 15).