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

F/SER31:CB
SERO-2024-02183

03/13/2025

Christy Fellas
Compliance Coordinator
Deepwater Horizon Program
Restoration Center, Office of Habitat Conservation
NOAA Fisheries Service
263 13th Avenue South
St. Petersburg, Florida 33701

Ref.: CPRA Project Number BA-0214, Louisiana Trustee Implementation Group, Grand Isle State Park Improvement, Grand Isle, Jefferson Parish, Louisiana.

Dear Christy Fellas,

The enclosed Biological Opinion responds to your request for reinitiation of a consultation with us, the National Marine Fisheries Service (NMFS), pursuant to Section 7 of the Endangered Species Act of 1973, as amended (16 U.S.C. § 1531 et seq.) for the above referenced action. The Opinion has been given the NMFS tracking number SERO-2024-02183. Please use the NMFS tracking number in all future correspondence related to this action.

This Opinion is a reinitiation of a previous Opinion (SERO-2018-00147; issued October 23, 2019; “2019 Opinion”), for the proposed reconstruction of a public recreational fishing pier funded by National Oceanic and Atmospheric Administration Restoration Center (NOAA RC). Since the issuance of the 2019 Opinion, the proposed design plan for the pier has been modified, including the proposed dimensions, location, and construction materials. The enclosed Opinion considers the effects of the NOAA RC’s proposal to fund the reconstruction of a recreational fishing pier by the Louisiana Trustee Implementation Group in conjunction with the Louisiana Coastal Protection and Restoration Authority (the applicant), in Grand Isle, Jefferson Parish, Louisiana on the following listed species: green sea turtle (North Atlantic DPS), Kemp’s ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray. The Opinion is based on information provided by the NOAA RC, the applicant, and the published literature cited within. NMFS concludes that the proposed action is likely to adversely affect, but is not likely to jeopardize the continued existence of, the green sea turtle (North Atlantic DPS), Kemp’s ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray.

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

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

This new opinion (SERO-2024-02183), including the Incidental Take Statement, Reasonable and Prudent Measures, and Terms and Conditions replaces and supersedes the 2019 Opinion.
We look forward to further cooperation with you on other projects to ensure the conservation of our threatened and endangered marine species and critical habitat. If you have any questions regarding this consultation, please contact Christopher Bond, OAI Contractor and Consultation Biologist supporting NMFS and OPR by email at chris.bond@noaa.gov.

Sincerely,

AMENDOLA.KIMBERLY.BARBARA.1365830769
Digitally signed by AMENDOLA.KIMBERLY.BARBARA.1365830769
Date: 2025.03.13 14:24:31 -04'00'

for Andrew J. Strelcheck
Regional Administrator

Enclosure:
NMFS Biological Opinion SERO-2024-02183
cc: christina.fellas@noaa.gov
nmfs.ser.esa.consultations@noaa.gov
File: 1514-22.c

Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	National Oceanic and Atmospheric Administration Restoration Center Permit number: BA-0214
Applicant:	Louisiana Trustee Implementation Group
Activity:	Grand Isle State Park Improvement project
Location:	Grand Isle, Jefferson Parish, Louisiana
Consulting Agency:	National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida NMFS Tracking Number: SERO-2024-02183

Approved by:

AMENDOLA.KIMBERLY.BARBARA.1365830769
Digitally signed by AMENDOLA.KIMBERLY.BARBARA.1365830769
Date: 2025.03.13 14:24:56 -04'00' for

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

Date Issued: 03/13/2025

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACRONYMS, ABBREVIATIONS, AND UNITS OF MEASURE

ac	acre(s)
°C	degrees Celsius
CFR	Code of Federal Regulations
cm	centimeter(s)
DPS	Distinct Population Segment
ECO	Environmental Consultation Organizer
EFH	Essential Fish Habitat
ESA	Endangered Species Act of 1973, as amended (16 U.S.C. § 1531 et seq.)
°F	degrees Fahrenheit
ft	foot/feet
FR	Federal Register
ft^2	square foot/feet
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Florida Fish and Wildlife Research Institute
in	inch(es)
IPCC	Intergovernmental Panel on Climate Change
km	kilometer(s)
lin ft	linear foot/feet
m	meter(s)
MHW	Mean High Water
mi	mile(s)
mi^2	square mile(s)
MLLW	Mean Lower Low Water
MMPA	Marine Mammal Protection Act
MMF	Marine Megafauna Foundation
MSA	Magnuson-Stevens Fishery Conservation and Management Act
N/A	not applicable
NAD 83	North American Datum of 1983
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
Opinion	Biological Opinion, Conference Biological Opinion, or Draft Biological Opinion
PRM	post release mortality
SERO PRD	NMFS Southeast Regional Office, Protected Resources Division
SAV	Submerged Aquatic Vegetation
SSRIT	Smalltooth Sawfish Recovery Implementation Team
STSSN	Sea Turtle Stranding and Salvage Network
U.S.	United States of America
USACE	United States Army Corps of Engineers
USFWS	United States Fish and Wildlife Service

1 INTRODUCTION

1.1 Overview

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

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

This document represents NMFS' Opinion based on our review of potential effects of the NOAA RC's proposal to fund the reconstruction of a recreational fishing pier by the Louisiana Trustee Implementation Group, in conjunction with the Louisiana Coastal Protection and Restoration Authority (the applicant) in Grand Isle, Jefferson Parish, Louisiana, on the following listed species: green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray. Our Opinion is based on information provided by the NOAA RC, the applicant, and the published literature cited within.

Updates to the regulations governing interagency consultation (50 CFR part 402) were effective on May 6, 2024 (89 FR 24268). We are applying the updated regulations to this consultation. The 2024 regulatory changes, like those from 2019, were intended to improve and clarify the consultation process, and, with one exception from 2024 (offsetting reasonable and prudent measures), were not intended to result in changes to the Services' existing practice in implementing section 7(a)(2) of the Act (89 FR 24268; 84 FR 45015). We have considered the prior rules and affirm that the substantive analysis and conclusions articulated in this Opinion and incidental take statement would not have been any different under the 2019 regulations or pre-2019 regulations

The Gulf of Mexico was renamed the Gulf of America pursuant to Executive Order 14172, and Secretary of the Interior Order No. 3423. All geographical references to the Gulf of America or "the Gulf" in this Opinion refer to the same body of water formerly known as the Gulf of Mexico.

1.2 Consultation History

The following is the consultation history for the NMFS ECO tracking number SERO-2024-02183 Grand Isle State Park Improvement Project.

On October 23, 2019, NMFS issued the 2019 Opinion for NOAA RC's proposal to fund the Grand Isle State Park Improvement Project in Grand Isle, Jefferson Parish, Louisiana. The 2019 Opinion determined that the proposed action was likely to adversely affect green sea turtle (North Atlantic and South Atlantic DPSs), Kemp's ridley sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS), but is not likely to jeopardize the continued existence of these species. The 2019 Opinion issued an Incidental Take Statement for green (North and South Atlantic DPSs), Kemp's ridley, and loggerhead sea turtles. The 2019 Opinion did not consider effects of the proposed project on giant manta ray.

On August 30, 2024, we received a written request from the NOAA RC for reinitiation of formal consultation under section 7 of the ESA on the remaining portion of the Grand Isle State Park Improvement Project (i.e., reconstruction of a public recreational fishing pier) due to modifications of the proposed dimensions, location, and construction materials for the pier.

On November 21, 2024, we requested additional information related to the proposed in-water pier lighting and other aspects of the proposed pier construction.

We received a final response on January 2, 2025, and initiated formal consultation that day.

This new opinion (SERO-2024-02183), including the Incidental Take Statement, Reasonable and Prudent Measures, and Terms and Conditions replaces and supersedes the 2019 Opinion.

2 PROPOSED ACTION

2.1 Project Details

2.1.1 Project Description

The NOAA RC proposes to fund the reconstruction of a public recreational fishing pier by the Louisiana Trustee Implementation Group in conjunction with the Louisiana CPRA. The proposed action involves the demolition of an existing, storm-damaged pier, and constructing a new upgraded pier to improve fishing access. The project will provide improved fishing and recreational use of Grand Isle State Park. The existing wooden pier was previously damaged from Hurricane Ida. Instead of repairing and extending the pier using the existing footprint, as proposed in the 2019 Opinion, the applicant proposes to replace it in a new footprint adjacent to the existing one using concrete instead of timber, which could be more resistant to damage from future storm events.

The proposed action is located within Grand Isle on the northeastern side of the island. The existing, damaged T-shaped fishing pier extends 300 ft southeast into the Gulf of America. The pier was severely damaged by Hurricane Ida and is currently closed to the public. Additionally, sediment build up around the pier has degraded fishing access by reducing the water depth around the pier. The proposed project involves the removal of the damaged pier and the construction of a new 1,200-ft-long concrete fishing pier located immediately adjacent to the footprint of the original pier. The completed replacement pier will be approximately 13,000 ft^2 in size with built-in benches and will sit 19 ft above the mean high-water line. The new pier will operate from 7:00 am to 9:00 pm, 365 days a year. Approximately 100 people will visit the pier to engage in daily fishing activities. The new pier will feature one fish cleaning station located on the land-based portion of the pier with ample garbage receptacles for anglers to dispose of fish carcasses. Grand Isle State Park officials will be responsible for disposing of this waste. The new pier will feature sea turtle friendly lighting on the pier walkway and terminus, as well as fish-attracting lights beneath the pier. Specifically, the lighting beneath the pier will be installed with a cover and pointed toward the shore to reduce Gulf-facing lighting that could attract sea turtles to the pier. Grand Isle State Park officials will be on-site during the pier's operational hours.

The existing damaged pier will be demolished by removing the existing timber piles. These piles will be cut or broken below the mudline. All piles on land will be pulled, if possible. A land-based crane and dumpster will be used for removing land piles. A barge-mounted crane and dumpster will be used to remove in-water piles. Demolition of the existing pier is estimated to take 18 months. The new pier will be supported by (60) 16 in diameter precast, prestressed concrete piles. Each pile measures 120 ft in length and will be spaced 15 ft apart (**Table 1**). On land, support pile installation includes (72) 14 in diameter precast, prestressed concrete piles measuring 60 ft each and will be spaced 8 ft apart. In-water pile installation will occur over several construction phases. Piles will be installed via impact hammer using standard equipment (crane, boom, pile hammer, pile gate, pile monkey, set of leads, and helmet). The crane will be barge-mounted and will access the project location via an access channel located immediately adjacent to the northern side of the proposed pier location. The access channel will be dredged via a bucket dredge method, which is estimated to take 3 months to complete. It is estimated that the dredge depth will be slightly greater than the 6-8 ft depth that the barge will require to transit through. The size of the dredged access channel is estimated to be 80 ft wide by 700 ft long with a maximum depth of 6 ft near the shore and decreasing toward the Gulf. Approximately 6,000 yd^3 of material will be dredged from the access channel and placed adjacent to the channel for temporary storage. In-water pile driving activities will take approximately 10 consecutive days to complete. The estimated total time to complete all in-water work (demolition of the existing pier, access channel dredging, in-water pile driving, and lighting installation) is approximately 11 months.

Table 1. In-Water Pile Installation Information for the Grand Isle State Park Improvement Project.

Pile Type	Pile Diameter (in)	Number of Piles	Installation Method	Average Number of Strikes per Pile (using impact hammer)	Maximum Number of Piles per Day	Confirmed Space or Open Water
Concrete	16	60	Impact	1200	6	Open Water

2.1.2 Mitigation Measures

To minimize potential impacts to ESA-listed species, the NOAA RC will ensure the following conditions will be followed by the applicant during construction:

- The applicant will adhere to and implement NMFS SERO's Protected Species Construction Conditions (https://media.fisheries.noaa.gov/2021-06/Protected_Species_Construction_Conditions_1.pdf?null), which require that operations of moving equipment shall cease if a protected species is observed within 150 ft of operations. Activities shall not resume until the protected species has departed the project area of its own volition (e.g., the species was observed departing or 20 minutes have passed since the animal was last seen in the area).
- The applicant will implement and adhere to NMFS SERO's *Vessel Strike Avoidance Measures* (NMFS 2021). All construction personnel must watch for and avoid collision with listed species.
- The applicant will implement and adhere to NMFS SERO's *Measures for Reducing the Entrapment Risk to Protected Species* (NMFS 2012)
- State Park staff will complete training on landing, handling, and de-hooking sea turtles.
- State Park staff will have ready access to sea turtle landing and de-hooking equipment (long-handled nets, pliers, etc.), as well as current contact information for local sea turtle rescue and rehabilitation facilities.
- All in-water work will be conducted during daylight hours to allow for the detection of any ESA-listed species that may enter the construction zones.
- Use of the existing parking lot or adjacent paved surface for delivery and storage of the majority of construction material and equipment should be used, if available.
- Prior to the onset of construction activities, the applicant or designated agent will conduct a meeting with all construction staff to discuss identification of ESA-listed species, their protected status, what to do if any are observed within the project area, and applicable penalties that may be imposed if State or Federal regulations are violated. All personnel shall be advised that there are civil and criminal penalties for harming, harassing, or killing ESA-listed species or marine mammals.
- When in-water project construction takes place from floating equipment (e.g., barge), prop or wheel-washing is prohibited.
- The new pier will feature sea turtle friendly lighting on the pier, and artificial lighting beneath the pier. Specifically, the lighting beneath the pier will be installed with a cover

and pointed toward the shore to reduce Gulf-facing lighting that could attract sea turtles to the pier.

- The applicant shall report any injury to any ESA-listed species occurring during the construction phase of the project immediately to:
  - NMFS SERO PRD via the NMFS SERO Endangered Species Take Report Form (https://forms.gle/85fP2da4Ds9jEL829). The applicant will include the SERO ECO tracking number in all correspondence.

2.1.3 Best Practices

To minimize potential impacts to ESA-listed species, the NOAA RC will ensure the following conditions will be followed by the applicant post-construction:

- Quarterly angler surveys will be conducted at the pier and will include questions on sea turtle encounters at the pier.
- Reports detailing available data on captures of sea turtles from the fishing pier will be submitted to the NMFS at the end of each calendar year in which such a capture is reported.
- Within 24 hours of any reported capture, entanglement, stranding, or other take, Gulf Island State Park staff must notify the NMFS SERO by email (takereport.nmfsser@noaa.gov).
- Monofilament recycling bins and trash receptacles will be placed on the fishing pier and maintained/emptied regularly to reduce the probability of trash and debris entering the water.
- Fish cleaning stations will be built away from the water, on the land-based portion of the pier, with ample garbage disposal facilities to avoid attracting sea turtles.
- The applicants will post signage that will ask anglers not to dispose of fish carcasses or debris in water.
- Annual underwater fishing debris cleanup around the pier would be conducted to remove any fishing line, nets, and other debris and trash from the water. Reports describing the results of each cleaning event will be submitted to the NMFS.
- Upon completion of the pier, the applicant will install NMFS-approved educational signs in visible locations at least at the entrance to and the terminal end of the pier, alerting users of listed species in the area. The applicant will post at the pier the following signs, in both English and Spanish, which are available for download at the following website: https://www.fisheries.noaa.gov/southeast/consultations/protected-species-educational-signs
  - *'Save Dolphins, Sea Turtles, Sawfish, and Manta Ray' signs.*
  - *'Do Not Catch or Harass Sea Turtles'*

2.2 Action Area

The project site is located at 29.25992° N, 89.95023° W (NAD 83) in Grand Isle, Jefferson Parish, Louisiana. The project site is located within the Grand Isle State Park along the northeastern shoreline of Grand Isle immediately adjacent to the Gulf of America. The existing 300-ft-long pier was severely damaged by Hurricane Ida and is currently closed to the public.

The action area is defined by regulation as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR 402.02). For the purposes of this federal action, the action area includes the nearshore areas in which construction will take place. The boundaries of the action area extends through a 1,120.3 ft (341.47 m) buffer around the locations where the old fishing pier demolition and the construction activities for the replacement pier will occur. This buffer encompasses the area where behavioral effects to the giant manta ray and sea turtles related to noise from pile driving may occur, as well as the general area accessible by recreational anglers fishing from the pier (i.e. casting distance or approximately 200 ft). The total acreage within this defined action area is approximately 88.7 ac (**Figure 1**). Substrate within the action area consists primarily of sand and silt soils with water depths ranging between 0 ft and 10 ft. There are no corals, mangroves, SAV, or designated critical habitat within the proposed action area.

Figure 1. The Action Area for the Grand Isle State Park Improvement Project.

3 EFFECTS DETERMINATIONS

Please note the following abbreviations are only used in **Table 1** and **Table 2** and are not, therefore, included in the list of acronyms: E = endangered; T = threatened; P = Proposed; LAA = likely to adversely affect; NLAA = may affect, not likely to adversely affect; NE = no effect.

3.1 Effects Determinations for ESA-Listed Species

3.1.1 Agency Effects Determinations

We have assessed the ESA-listed species that may be present in the action area and our determination of the project's potential effects is shown in **Table 1** below.

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

Species (DPS)	ESA Listing Status	Listing Rule/Date	Most Recent Recovery Plan (or Outline) Date	NOAA RC Effect Determination	NMFS Effect Determination
Sea Turtles[1]					
Green sea turtle (North Atlantic DPS)	T	81 FR 20057/ April 6, 2016	October 1991	LAA	LAA
Kemp's ridley sea turtle	E	35 FR 18319/ December 2, 1970	September 2011	LAA	LAA
Loggerhead sea turtle (Northwest Atlantic DPS)	T	76 FR 58868/ September 22, 2011	December 2008	LAA	LAA
Fishes					
Giant manta ray	T	83 FR 2916/ January 22, 2018	2019 (Outline)	LAA	LAA

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

We have determined that the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray are likely to be adversely affected by the proposed action and thus require further analysis. We provide greater detail on the potential effects to these species from the proposed action in the Effects of the Action (Section 6) and whether those effects, when considered in the context of the Status of the Species (Section 4), the Environmental Baseline (Section 5), and the Cumulative Effects (Section 7), are likely to likely to jeopardize the continued existence of these ESA-listed species in the wild.

[1] The 2019 Opinion considered effects of the proposed pier replacement on the South Atlantic DPS of green sea turtle. Limited information previously indicated that benthic juveniles from both the North Atlantic and South Atlantic DPSs may be found in waters off the mainland United States. However, additional research has determined that juveniles from the South Atlantic DPS are not likely to occur in these waters, including the action area for this project.

The Grand Isle State Park Fishing Pier is located in the Gulf of America-facing waters of Zone 12, a statistical subarea used when reporting commercial fishing data. Zone 12 extends from 29º to 30º North latitude and from 88° to 90° West longitude from Biloxi, Mississippi west to New Orleans, Louisiana. To help determine which sea turtle species are likely to occur within the action area, we reviewed the available years of STSSN offshore stranding data (i.e., stranding data for all areas outside of protected waters for the years 2007-2016) for Zone 12 (**Table 3**). Given the near-total lack of past reporting of fishing pier captures in the State of Louisiana, we believe that the available capture/reporting data from Mississippi is the best surrogate for estimating future sea turtle captures and reporting from the Grand Isle State Park Fishing Pier. This dataset is the best available in terms of STSSN data that has been processed to the granularity necessary to accurately determine strandings due to entanglement and hook-and-line captures from a structure. While more recent STSSN data is available, it has not been processed or validated to a level that accurately informs the mortality calculations we conduct in the Effects of the Action (Section 6). Further, we conducted an analysis using the raw updated data to determine whether take calculations would be significantly different from those using the available processed 10-year dataset and determined that use of the available dataset is still appropriate.

The Grand Island State Park fishing pier is ocean-facing; however, we chose to review both inshore and offshore data for Zone 12. The Mississippi Stranding Network codes the data for the majority of fishing piers in Zone 12 as inshore data; therefore, using both inshore and offshore data for Zone 12 ensures we capture all strandings at fishing piers within Zone 12 in our analysis. Based on the 2007-2016 STSSN dataset, we believe only green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, and loggerhead sea turtle (Northwest Atlantic DPS) may be affected by construction effects as well as recreational fishing that will occur at the pier upon completion of the proposed action (Table 3). While leatherback sea turtle is represented in the data, we do not believe this species will be in the action area or caught on or entangled in recreational hook and line gear used at the pier. As noted in Table 3, the STSSN record for a single leatherback sea turtle is the result of an unknown activity. Further, leatherback sea turtles tend to be pelagic feeders, feeding on jellyfish and not baits typically fished from piers.

Table 3. Summary of STSSN Inshore and Offshore Data for Zone 12 (2007-2016)

Species	Number of Sea Known Turtles Stranded or Salvaged (All Activities)	Number of Known Gear Entanglements	Number of Known Recreational Hook-and-line Captures
Green sea turtle	17	0	5
Kemp's ridley sea turtle	1,490	12	638
Leatherback sea turtle	1	0	0
Loggerhead sea turtle	44	0	7
Total	**1,552**	**12**	**650**

Giant manta ray are also prone to interactions with recreational fishing gear via foul-hooking (i.e., when an animal is hooked anywhere on the body without having taken the bait in its mouth) particularly at fishing structures that are ocean-facing or located in or near inlet/passes. Based on the best available data, we believe that giant manta may be found in the action area and are likely to be adversely affected by recreational hook-and-line interactions upon the completion of the proposed action.

We have determined that green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray are likely to be adversely affected by the proposed action and thus require further analysis. We provide greater detail on the potential effects to these species from the proposed action in the Effects of the Action (Section 6) and whether those effects, when considered in the context of the Status of the Species (Section 4), the Environmental Baseline (Section 5), and the Cumulative Effects (Section 7), are likely to likely to jeopardize the continued existence of these ESA-listed species in the wild.

4 STATUS OF ESA-LISTED SPECIES CONSIDERED FOR FURTHER ANALYSIS

4.1 Rangewide Status of the Species Considered for Further Analysis

4.2 Overview of Status of Sea Turtles

There are five species of sea turtles (green, hawksbill, Kemp's ridley, leatherback, and loggerhead) that travel widely throughout the South Atlantic, Gulf of America and the Caribbean. These species are highly migratory and therefore could occur within the action area. Section 4.2.1 will address the general threats that confront all sea turtle species. The remainder of Section 4.2 (Sections 4.2.2-4.2.6) will address information on the distribution, life history, population structure, abundance, population trends, and unique threats to the species of sea turtles that are likely to be adversely affected by the proposed action, namely, green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS).

4.2.1 General Threats Faced by All Sea Turtle Species

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

4.2.1.1 Fisheries

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

In addition to domestic fisheries, sea turtles are subject to direct as well as incidental capture in numerous foreign fisheries, further impeding the ability of sea turtles to survive and recover on a global scale. For example, pelagic stage sea turtles, especially loggerheads and leatherbacks, circumnavigating the Atlantic are susceptible to international longline fisheries including the Azorean, Spanish, and various other fleets (Aguilar et al. 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.

4.2.1.2 Non-Fishery In-Water Activities

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

4.2.1.3 Coastal Development and Erosion Control

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

4.2.1.4 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 oil rig affected sea turtles in the Gulf of America. An assessment has been completed on the injury to Gulf of America 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 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 lost, abandoned or discarded fishing gear) as they feed along oceanographic fronts where debris and their natural food items converge. Marine debris can cause significant habitat destruction from derelict vessels, further exacerbated by tropical storms moving debris and scouring and destroying corals and seagrass beds, for instance. Sea turtles that spend significant portions of their lives in the pelagic environment (i.e., juvenile loggerheads, and juvenile green turtles) are especially susceptible to threats from entanglement in marine debris when they return to coastal waters to breed and nest.

4.2.1.5 Climate Change

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

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

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

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

4.2.1.6 Other Threats

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

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

4.2.2 Status of Green Sea Turtle – North Atlantic DPS

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 distinct population segments (DPSs) (81 FR 20057 2016) (Figure 2). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific DPSs were listed as threatened. Only individuals from the South

Atlantic DPS and North Atlantic DPS may occur in waters under the purview of the NMFS SE Region, with South Atlantic DPS individuals only expected to occur in the U.S. Caribbean.

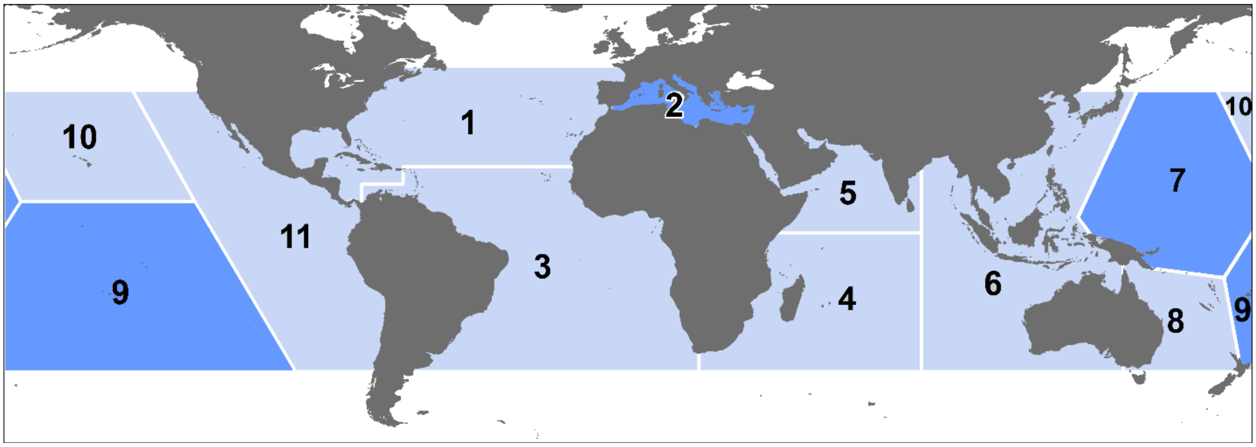

Figure 2. Threatened (light) and Endangered (dark) Green Sea Turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and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

Species Description and Distribution

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

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

Differences in mitochondrial deoxyribonucleic acid (DNA) properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; FitzSimmons et al. 2006). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Limited early information indicated that within U.S. waters benthic juveniles from both the North Atlantic and South Atlantic DPSs may be found on foraging grounds. Two small-scale studies provided an insight into the possible degree of mixing on the foraging grounds. An analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of America) found approximately 4% of individuals came from nesting stocks in the South Atlantic DPS (specifically Suriname, Aves Island, Brazil, Ascension Island, and Guinea Bissau) (Foley et al. 2007). On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the Aves Island/Suriname nesting assemblage, which is part of the South Atlantic DPS (Bass and Witzell 2000). Available information on green turtle migratory behavior indicates that long distance dispersal is only seen for juvenile turtles. This suggests that larger adult-sized turtles return to forage within the region of their natal rookeries, thereby limiting the potential for gene flow across larger scales (Monzón-Argüello et al. 2010). However, with additional research it has been determined that South Atlantic juveniles are not likely to be occurring in U.S. mainland coastal waters in anything more than negligible numbers. Jensen et al. (2013) indicated that the earlier studies might represent a statistical artifact as they lack sufficient precision, with error intervals that span zero. More recent studies with better rookery baseline representation found negligible (<1%) contributions from the South Atlantic DPS among Texas and Florida GoM juvenile green turtle assemblages (Shamblin et al. 2016, 2018). Finally, an as-yet published genetic analysis of samples from various coastal areas in the Gulf of America and Atlantic has now solidified the conclusion that South Atlantic juveniles represent at best a negligible number of individuals in mainland United States waters (Peter Dutton, SWFSC, pers. comm. April 2022). Therefore, we will not consider South Atlantic DPS individuals when conducting consultations for projects in the waters off the mainland United States.

The North Atlantic DPS boundary is illustrated in Figure 2. Four regions support nesting concentrations of particular interest in the North Atlantic DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

The complete nesting range of North Atlantic DPS green sea turtles within the southeastern United States includes sandy beaches between Texas and North Carolina, as well as Puerto Rico (Dow et al. 2007; NMFS and USFWS 1991a). 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 America 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 America off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1992; Wershoven and Wershoven 1992). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as

Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatán Peninsula.

Life History Information

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

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

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

Status and Population Dynamics

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

The North Atlantic DPS is the largest of the 11 green turtle DPSs, with an estimated nester abundance of over 167,000 adult females from 73 nesting sites. Overall, this DPS is also the most data rich. Eight of the sites have high levels of abundance (i.e., >1000 nesters), located in Costa Rica, Cuba, Mexico, and Florida.

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

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

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

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the

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

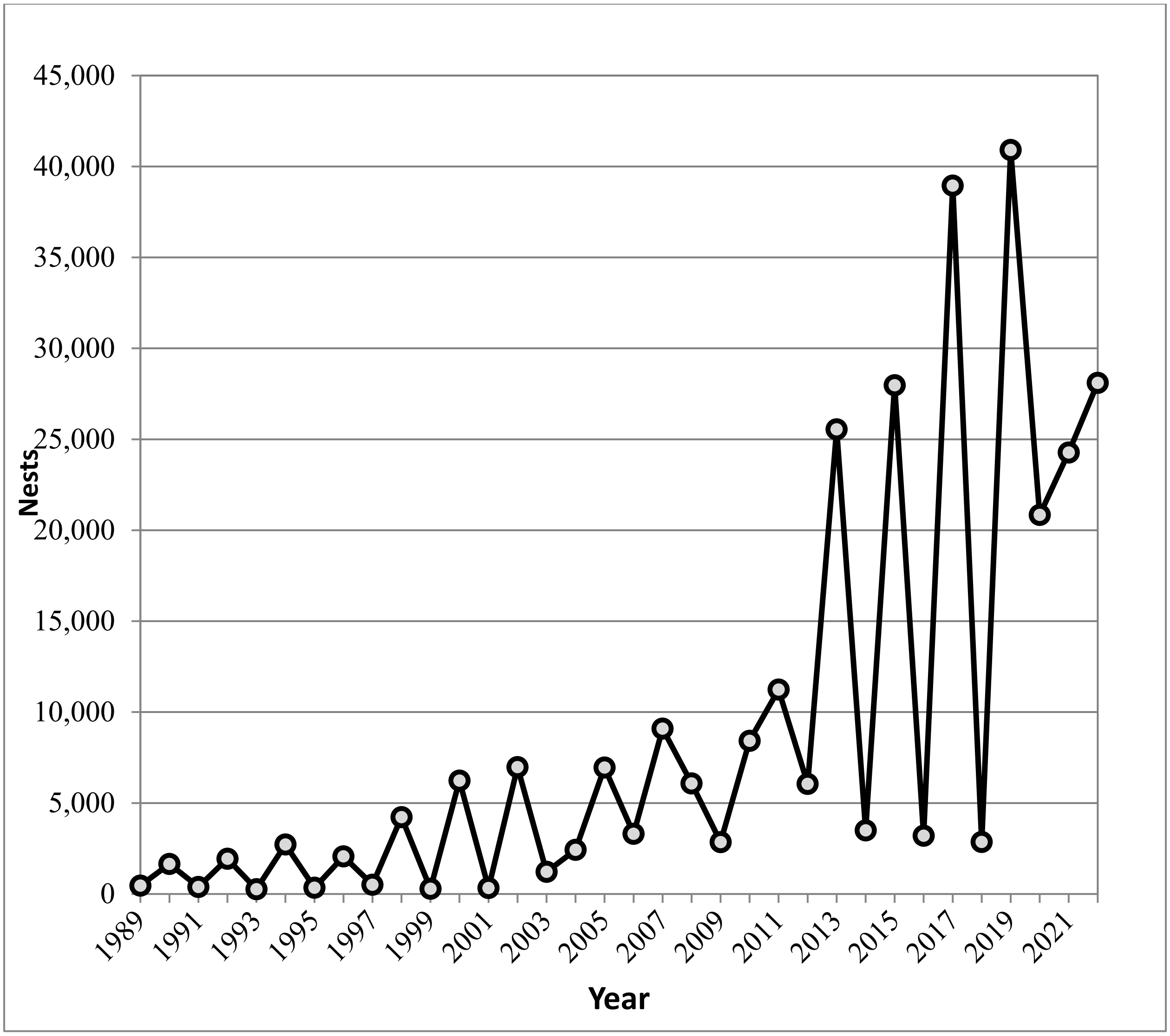

Figure 3. Green Sea Turtle Nesting at Florida Index Beaches Since 1989.

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

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.2.1.

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

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

Whereas oil spill impacts are discussed generally for all species in Section 4.2.1, specific impacts of the Deepwater Horizon (DWH) spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juveniles greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (DWH Trustees 2015). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil or dispersants, and loss of foraging resources, which could lead to compromised growth 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 America, they have a widespread distribution throughout the entire Gulf of America, Caribbean, and Atlantic, and the proportion of the population using the northern Gulf of America at any given time is relatively low. Although it is known that adverse impacts occurred and numbers of animals in the Gulf of America were reduced as a result of the Deepwater Horizon oil spill of 2010 (DWH), the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event, as well as the impacts being primarily to smaller juveniles (lower reproductive value than adults and large juveniles), reduces the impact to the overall population. It is unclear what impact these losses may have caused on a population level, but it is not expected to have had a large impact on the population trajectory moving forward. However, recovery of green turtle numbers equivalent to what was lost in the northern Gulf of America as a result of the spill will likely take decades of sustained efforts to reduce the existing threats and enhance survivorship of multiple life stages (DWH Trustees 2015).

4.2.3 Status of Kemp's Ridley Sea Turtle

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

Species Description and Distribution

The Kemp's ridley sea turtle is the smallest of all sea turtles. Adults generally weigh less than 100 lb (45 kg) and have a carapace length of around 2.1 ft (65 cm). Adult Kemp's ridley shells are almost as wide as they are long. Coloration changes significantly during development from the grey-black dorsum and plastron of hatchlings, a grey-black dorsum with a yellowish-white plastron as post-pelagic juveniles, and then to the lighter grey-olive carapace and cream-white or yellowish plastron of adults. There are 2 pairs of prefrontal scales on the head, 5 vertebral scutes, usually 5 pairs of costal scutes, and generally 12 pairs of marginal scutes on the carapace. In each bridge adjoining the plastron to the carapace, there are 4 scutes, each of which is perforated by a pore.

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

The primary range of Kemp's ridley sea turtles is within the Gulf of America basin, though they also occur in coastal and offshore waters of the U.S. Atlantic Ocean. Juvenile Kemp's ridley sea turtles, possibly carried by oceanic currents, have been recorded as far north as Nova Scotia. Historic records indicate a nesting range from Mustang Island, Texas, in the north to Veracruz, Mexico, in the south. Kemp's ridley sea turtles have recently been nesting along the Atlantic

Coast of the United States, with nests recorded from beaches in Florida, Georgia, and the Carolinas. In 2012, the first Kemp's ridley sea turtle nest was recorded in Virginia. The Kemp's ridley nesting population had been exponentially increasing prior to the recent low nesting years, which may indicate that the population had been experiencing a similar increase. Additional nesting data in the coming years will be required to determine what the recent nesting decline means for the population trajectory.

Life History Information

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

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

Population Dynamics

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

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

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

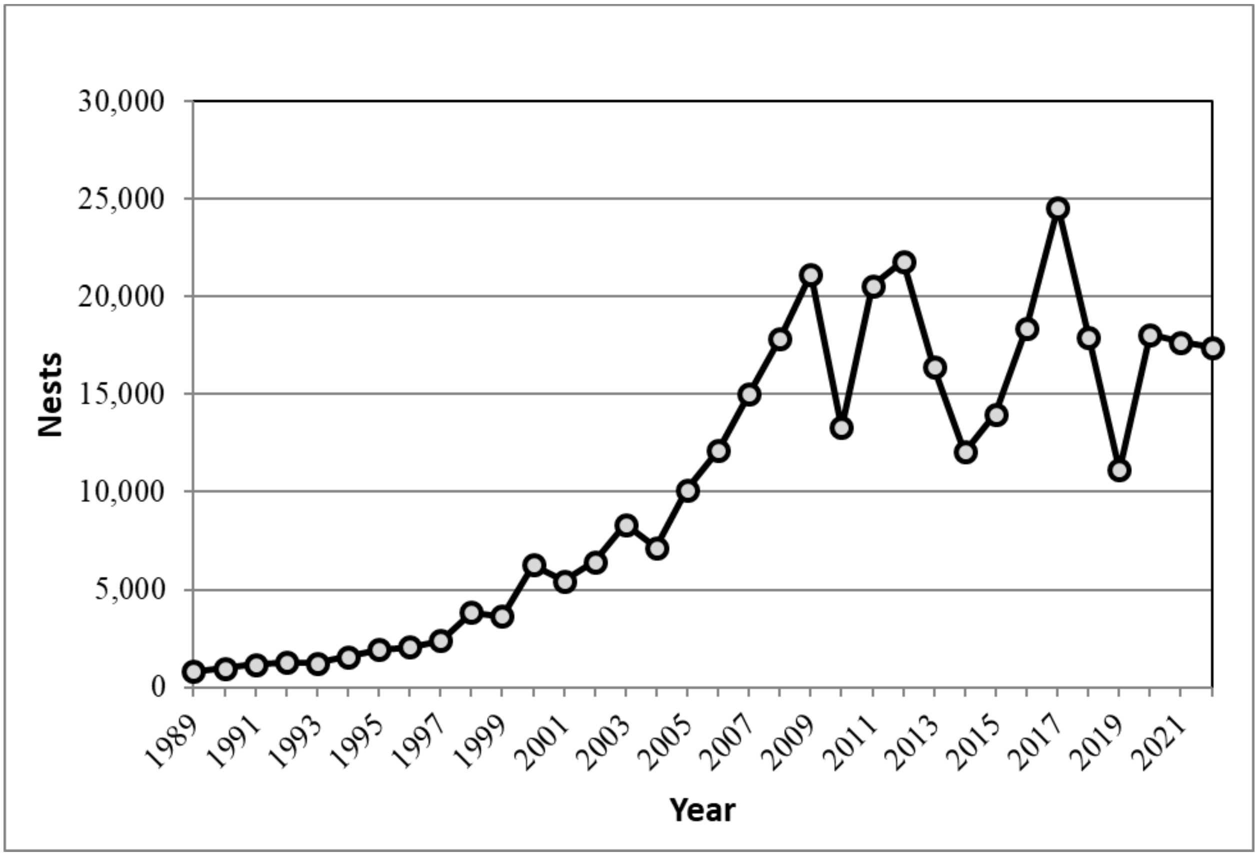

Figure 4. Kemp's Ridley Nest Totals from Mexican Beaches (M. Barnette pers. comm. to Jaime Pena, Gladys Porter Zoo, 2019 and CONAMP Data 2020-2022)

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

Threats

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

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

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

In response to these strandings, and due to speculation that fishery interactions may be the cause, fishery observer effort was shifted to evaluate the inshore skimmer trawl fisheries beginning in 2012. During May-July of that year, observers reported 24 sea turtle interactions in the skimmer trawl fisheries. All but a single sea turtle were identified as Kemp's ridleys (1 sea turtle was an unidentified hardshell turtle). Encountered sea turtles were all very small juvenile specimens, ranging from 7.6-19.0 in (19.4-48.3 cm) curved carapace length (CCL). Subsequent years of observation noted additional captures in the skimmer trawl fisheries, including some mortalities. The small average size of encountered Kemp's ridleys introduces a potential conservation issue, as over 50% of these reported sea turtles could potentially pass through the maximum 4-in bar spacing of TEDs currently required in the shrimp fisheries. Due to this issue, a proposed 2012 rule to require 4-in bar spacing TEDs in the skimmer trawl fisheries (77 FR 27411) was not implemented. Following additional gear testing, however, we proposed a new rule in 2016 (81 FR 91097) to require TEDs with 3-in bar spacing for all vessels using skimmer trawls, pusher-head trawls, or wing nets. Ultimately, we published a final rule on December 20, 2019 (84 FR 70048), that requires all skimmer trawl vessels 40 feet and greater in length to use TEDs designed to exclude small sea turtles in their nets effective April 1, 2021. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of America may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 4.2.1, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles are considered here. Kemp's ridleys experienced the greatest negative impact stemming from the DWH oil spill event of any sea turtle species. Impacts to Kemp's ridley sea turtles occurred to offshore small juveniles, as well as large juveniles and adults. Loss of hatchling production resulting from injury to adult turtles was also estimated for this species. Injuries to adult turtles of other species, such as loggerheads, certainly would have resulted in unrealized nests and hatchlings to those species as well. Yet, the calculation of unrealized nests and hatchlings was limited to Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of America throughout their lives (DWH Trustees 2016).

A total of 217,000 small juvenile Kemp's ridleys (51.5% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. That means approximately half of all small juvenile Kemp's ridleys from the total population estimate of 430,000 oceanic small juveniles were exposed to oil. Furthermore, a large number of small juveniles were removed from the population, as up to 90,300 small juveniles Kemp's ridleys are estimated to have died as a direct result of the exposure. Therefore, as much as 20% of the small oceanic juveniles of this species were killed during that year. Impacts to large juveniles (>3 years old) and adults were also high. An estimated 21,990 such individuals were exposed to oil (about 22% of the total estimated population for those age classes); of those, 3,110 mortalities were estimated (or 3% of the population for those age classes). The loss of near-reproductive and reproductive-stage females would have contributed to some extent to the decline in total nesting abundance observed between 2011 and 2014. The estimated number of unrealized Kemp's ridley nests is between 1,300 and 2,000, which translates to between approximately 65,000 and 95,000 unrealized hatchlings (DWH Trustees 2016). This is a minimum estimate, however, because the sublethal effects of the DWH oil spill event on turtles, their prey, and their habitats might have delayed or reduced reproduction in subsequent years, which may have contributed substantially to additional nesting deficits observed following the DWH oil spill event. These sublethal effects could have slowed growth and maturation rates, increased remigration intervals, and decreased clutch frequency (number of nests per female per nesting season). The nature of the DWH oil spill event effect on reduced Kemp's ridley nesting abundance and associated hatchling production after 2010 requires further evaluation. It is clear that the DWH oil spill event resulted in large losses to the Kemp's ridley population across various age classes, and likely had an important population-level effect on the species. Still, we do not have a clear understanding of those impacts on the population trajectory for the species into the future.

4.2.4 Status of Loggerhead Sea Turtle – Northwest Atlantic DPS

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

This Opinion refers to the Northern Gulf of Mexico Recovery Unit (NGMRU) identified in the final rule listing for the nine distinct population segments of loggerhead sea turtle (78 FR 58868, Sept. 22, 2011) as the Northern Gulf of America Recovery Unit (NGARU). The geographical location of the recovery unit remains the same

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

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

The majority of loggerhead nesting occurs at the western rims of the Atlantic and Indian Oceans concentrated in the north and south temperate zones and subtropics (NRC 1990). For the 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 America, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997;

Addison and Morford 1996), off the southwestern coast of Cuba (Gavilan 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

Non-nesting, adult female loggerheads are reported throughout the U.S. Atlantic, Gulf of America, 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 America, and 5% in the western Gulf of America (TEWG 1998).

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

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

Life History Information
The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, which include the ecosystems those stages generally use: (1) egg (terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone- nearshore marine environment from the surface to the sea floor where water depths do not exceed 200 meters), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting female (terrestrial zone) (NMFS and USFWS 2008). Loggerheads are long-lived animals. They reach sexual maturity between 20-38 years of age, although age of maturity varies widely among

populations (Frazer and Ehrhart 1985; NMFS 2001). The annual mating season occurs from late March to early June, and female turtles lay eggs throughout the summer months. Females deposit an average of 4.1 nests within a nesting season (Murphy and Hopkins 1984), but an individual female only nests every 3.7 years on average (Tucker 2010). Each nest contains an average of 100-126 eggs (Dodd Jr. 1988) which incubate for 42-75 days before hatching (NMFS and USFWS 2008). Loggerhead hatchlings are 1.5-2 in long and weigh about 0.7 oz (20 g).

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

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

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

Status and Population Dynamics

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

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

Peninsular Florida Recovery Unit

The PFRU is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed an average of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (NMFS and USFWS 2008). The statewide estimated total for 2020 was 105,164 nests (FWRI nesting database).

In addition to the total nest count estimates, the FWRI uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. FWRI uses the standardized index survey data to analyze the nesting trends (Figure 5) (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). Since the beginning of the index program in 1989, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that was followed by a sharp decline over the subsequent 9 years. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represented a new record for loggerheads on the core index beaches. While nest numbers subsequently declined from the 2016 high FWRI noted that the 2007-2021 period represents a period of increase. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but non-significant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening CI. Nesting at the core index beaches declined in 2017 to 48,033, and rose again each year through 2020, reaching 53,443 nests, dipping back to 49,100 in 2021, and then in 2022 reaching the second-highest number since the survey began, with 62,396 nests. It is important to note that with the wide CIs and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.) it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani et al. 2019).

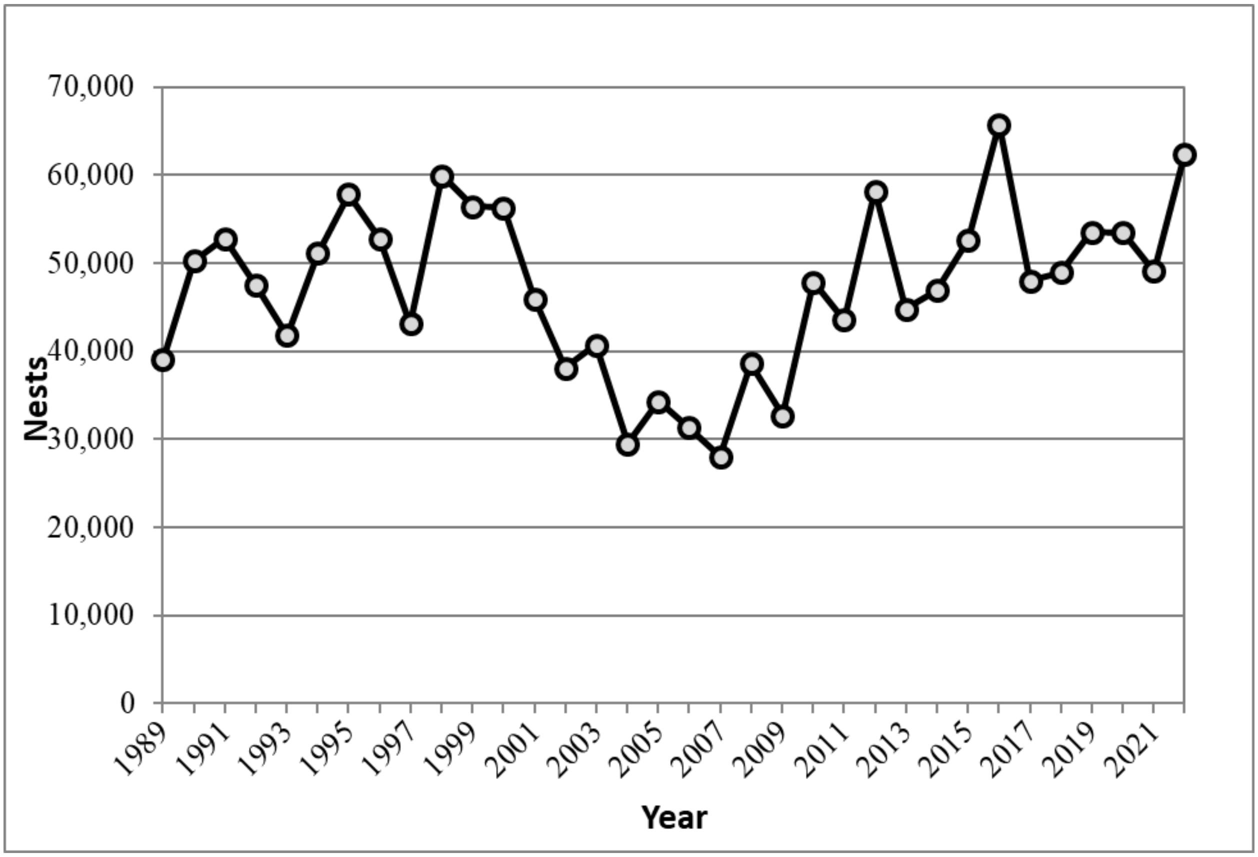

Figure 5. Loggerhead Sea Turtle Nesting at Florida Index Beaches Since 1989

<u>Northern Recovery Unit</u>

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

Data since that analysis (**Table 3**) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, GADNR press release, https://georgiawildlife.com/loggerhead-nest-season-begins-where-monitoring-began). South Carolina and North Carolina nesting have also begun to shift away from the past declining trend. Loggerhead nesting in Georgia, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016. Nesting in 2017 and 2018 declined relative to 2016, back to levels seen in 2013 to 2015, but then bounced back in 2019, breaking records for each of the three states and the overall recovery unit. Nesting in 2020 and 2021 declined from the 2019 records, but still remained high, representing the third and fourth highest total numbers for the NRU since 2008. In 2022 Georgia loggerhead nesting broke the record at 4,071, while South Carolina and North Carolina nesting were both at the second-highest level recorded.

Table 4. Total Number of NRU Loggerhead Nests (GADNR, SCDNR, and NCWRC Nesting Datasets Compiled at Seaturtle.org)

Year	Georgia	South Carolina	North Carolina	Totals
2008	1,649	4,500	841	**6,990**
2009	998	2,182	302	**3,482**
2010	1,760	3,141	856	**5,757**
2011	1,992	4,015	950	**6,957**
2012	2,241	4,615	1,074	**7,930**
2013	2,289	5,193	1,260	**8,742**
2014	1,196	2,083	542	**3,821**
2015	2,319	5,104	1,254	**8,677**
2016	3,265	6,443	1,612	**11,320**
2017	2,155	5,232	1,195	**8,582**
2018	1,735	2,762	765	**5,262**
2019	3,945	8,774	2,291	**15,010**
2020	2,786	5,551	1,335	**9,672**
2021	2,493	5,639	1,448	**9,580**
2022	4,071	7,970	1,906	**13,947**

In addition to the statewide nest counts, South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record. After another drop in 2018, a new record was set for the 2019 season, with a return to 2016 levels in 2020 and 2021 and then a rebound to the second highest level on record in 2022 (Figure 6.)

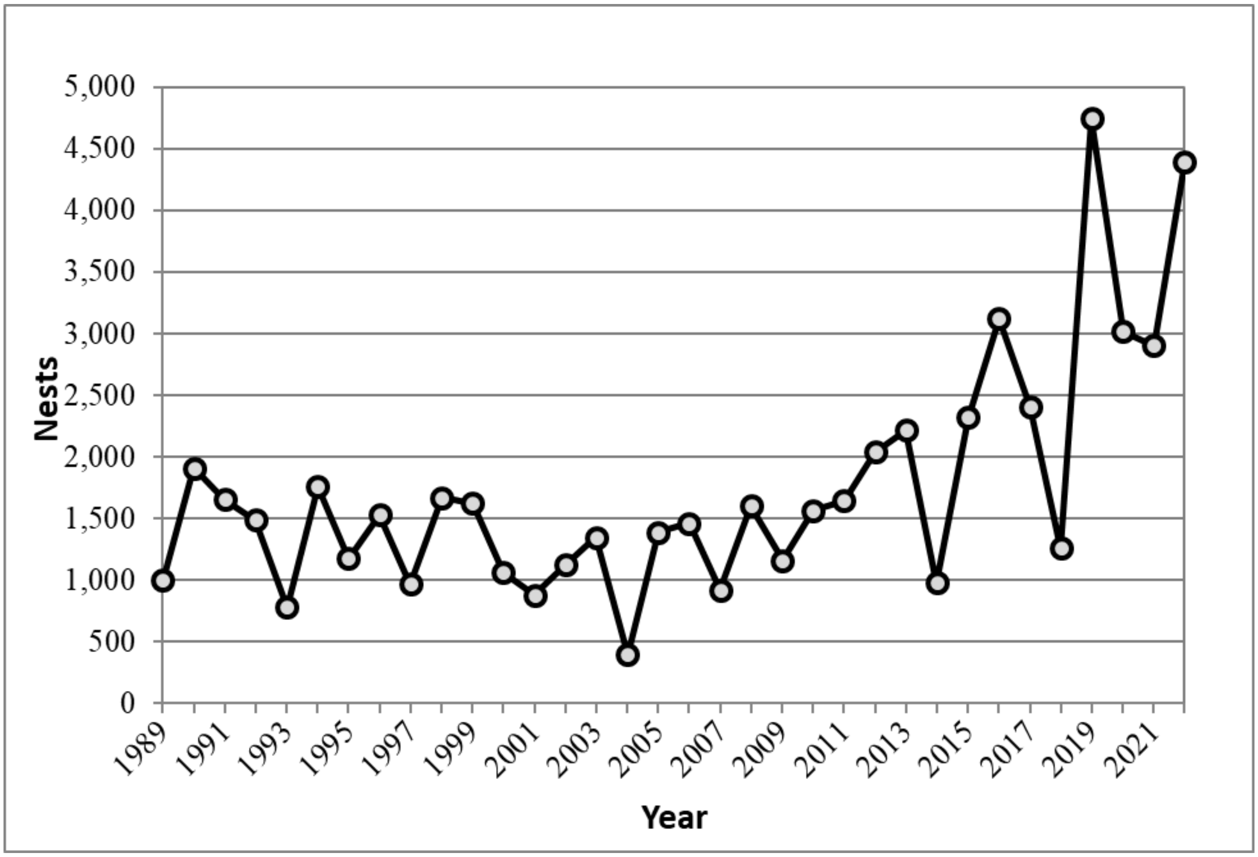

Figure 6. South Carolina Index Nesting Beach Counts for Loggerhead Sea Turtles (Data Provided by SCDNR)

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

The remaining 3 recovery units – Dry Tortugas (DTRU), Northern Gulf of America (NGARU), 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 NGARU 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 NGARU nesting, had shown a large increase in 2008, but then declined again in 2009 and 2010 before rising back to a level similar to the 2003-2007 average in 2011. From 1989-2018 the average number of NGARU nests annually on index beaches was 169 nests, with an average of 1100 counted in the statewide nesting counts (Ceriani et al. 2019). Nesting survey effort has been inconsistent among the GCRU nesting beaches, and no trend can be determined for this subpopulation (NMFS and USFWS 2008). Zurita et al. (2003) found a statistically significant increase in the number of nests on 7 of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. Nonetheless, nesting has declined since

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

In-water Trends

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

Population Estimate

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

Threats (Specific to Loggerhead Sea Turtles)

The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 4.2.1. Yet the impact of fishery interactions is a point of further emphasis for this species. The joint NMFS and USFWS Loggerhead Biological Review Team determined that the greatest threats to the Northwest Atlantic DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant et al. 2009).

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

While oil spill impacts are discussed generally for all species in Section 4.2.1, specific impacts of the DWH oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (DWH Trustees 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil or dispersants, and loss of foraging resources which could lead to compromised growth 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 DPS occurs on the Atlantic coast and, thus, loggerheads were impacted to a relatively lesser degree. However, it is likely that impacts to the NGARU of the Northwest Atlantic DPS would be proportionally much greater than the impacts occurring to other recovery units. Impacts to nesting and oiling effects on a large proportion of the NGARU recovery unit, especially mating and nesting adults likely had an impact on the NGARU. Based on the response injury evaluations for Florida Panhandle and Alabama nesting beaches, the DWH Trustees (2016) estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH oil spill response activities on nesting beaches. Although the long-term effects remain unknown, the DWH oil spill event impacts to the NGARU 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 America 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).

4.3 Giant Manta Ray

The giant manta ray is listed as a threatened species under the ESA (83 FR 2916, January 22, 2018). Critical habitat is not designated (84 FR 66652; December 5, 2019).

Species Description

The giant manta ray has a diamond-shaped body with wing-like pectoral fins; the distance over this wingspan is termed disc width (DW). It may be the largest living ray species, attaining a maximum size of 800 cm DW, with anecdotal reports up to 910 cm DW (Alava et al., 2002). There are two distinct color types: chevron and black (melanistic). Most of the chevron variants have a black dorsal surface and a white ventral surface with distinct patterns on the underside that can be used to identify individuals (Kitchen-Wheeler 2010; Deakos et al., 2011). While these markings are assumed to be permanent, there is some evidence that the pigmentation pattern of giant manta ray may actually change over the course of development (based on observation of two individuals in captivity), and thus caution may be warranted when using color markings for identification purposes in the wild. The black color variants are entirely black on the dorsal side and almost completely black on the ventral side, except for areas between the gill-slits and the abdominal area below the gill-slits (Kitchen-Wheeler 2013). They also have distinct spot patterns on their bellies that can be used to identify individuals (Kitchen-Wheeler, 2010; Deakos et al., 2011).

Habitat

The giant manta ray inhabits tropical, subtropical, and temperate bodies of water and is commonly found offshore, in oceanic waters, and near productive coastlines, with water temperatures generally between 20°C and 30°C (Marshall et al., 2009; Kashiwagi et al., 2011; Freedman and Roy 2012; Graham et al., 2012; Farmer et al., 2022). Manta rays are commonly seen in surface waters or cleaning in shallow coral reef habitats typically in tropical or subtropical regions (Couturier et al., 2012). The giant manta ray can exhibit diel patterns in habitat use, moving inshore during the day to clean and socialize in shallow waters (10-20 m), and then moving offshore at night to feed to depths of 1,000 meters (Hearn et al. 2014; Burgess 2016). The coastal vertical movements of giant manta rays maybe motivated by a combined foraging and thermal recovery strategy, whereby giant manta rays dived to forage on vertically migrating zooplankton at night and returned to surface waters (<2 m) to rewarm between dives (Andrzejaczek et al., 2021). In coastal areas, giant manta rays have been observed in shallow waters, sometimes less than 3 meters deep, in estuarine waters, near coastal inlets, with use of these shallow waters as potential nursery habitats (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al., 2015; Pate and Marshall 2020; Farmer et al., 2022).

Diet and Feeding

Giant manta rays primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderate sized fishes (Bigelow and Schroeder 1953; Rohner et al., 2017; Stewart et al., 2017). While there are no studies that compare diet requirements for the different life stages, recent studies suggest that both juveniles and adults may occupy the same habitats within a location and, thus, target the same prey (Stewart et al., 2016b, Graham et al., 2012). Manta rays have a complex depth profile of their foraging habitat (Andrzejaczek et al., 2021). Burgess et al., (2016) found that, on average, mesopelagic sources contributed 73% to the giant manta ray's diet, compared to 27% for surface zooplankton suggesting that giant manta rays may be supplementing their diet with opportunistic feeding in near-surface waters (Couturier et al., 2013; Burgess et al., 2016).

The feeding behaviors of manta species have also been studied to provide insight into their cognition and response to sensory stimuli. When feeding, groups of manta rays hold their cephalic fins in an "o" shape and open their mouths wide. They tend to swim at a speed around 30 pectoral fin beats per minute when feeding, which is almost twice as fast as they swim when being cleaned (Kitchen-Wheeler 2013). After collecting water with zooplankton in their mouths, manta rays use a transverse curtain on the roof of the mouth as a valve to hold the water in as the pharynx contracts during swallowing (Bigelow and Schroeder 1953). This movement of the pharynx pulls plankton towards the stomach when the gills are closed (Kitchen-Wheeler 2013). Intestinal eversion has also been observed, likely to clear the intestines of indigestible material and parasites. The positioning of the cephalic fins was found to be a good indicator of feeding motivation, triggered by underwater visual stimuli or olfactory stimuli / sense of smell (Ari and Correia 2008).

Reproduction

The giant manta ray is ovoviviparous and is thought to produce a single offspring per pregnancy after a gestation period of 12-13 months (Rambahiniarison et al., 2018). An average female produces 4–7 pups during its lifespan (Marshall et al., 2022).

Age at Maturity

Males mature at 350–400 cm DW and females mature at 380–500 cm DW (Stewart et al., 2018b). Female giant manta rays mature at 8.6 years of age, although first pregnancy may be delayed by up to 4 years depending on food availability (Rambahiniarison et al., 2018). Maximum age is estimated at 45 years and generation length is estimated to be between 20 years (J. Carlson unpublished) to 29 years (Marshall et al., 2022).

Habitat Use by Different Life Stages

Identifying potential manta ray important habitats, such as nursery, feeding and reproductive areas, especially in data limited regions such as the northwest Atlantic Ocean, is essential to conservation and recovery of this species. A recent study conducted off the Atlantic coast of central Florida provided evidence of reproductive habitat for manta rays (Pate 2024). Each spring, manta rays aggregate off the coast of central and northern Florida between Indian River County, Florida and the Florida/Georgia border. Pate (2024) documented numerous courtship and breaching events indicating that this area is potentially seasonal reproductive habitat for manta rays. This same study suggested that this area maybe important feeding habitat as feeding behaviors and prey species were also documented (Pate 2024). These initial observations warrant future study to determine the importance of this area to manta ray feeding and reproduction, as well as characterization of the environmental influences that affect manta ray presence and behavior.

Documenting juvenile nursery habitats is a priority in manta ray research and conservation (Stewart et al., 2018a), yet few have been described. Worldwide few nursery areas for manta rays have been described; however, two manta ray nursery habitats have been described in the U.S Atlantic and Gulf of America. Pate and Marshall (2020) described the nearshore area between St. Lucie Inlet and Boynton Beach Inlet in southeast Florida as nursery habitat for manta rays. Nearly all (98%) of manta rays observed by Pate and Marshall (2020) were juveniles and many showed high site fidelity to this nursery habitat. Observations of juvenile giant manta rays as far south as Miami suggest this nursery habitat may extend farther south (J. Pate, MMF, pers comm. to C. Horn, NMFS, June 4, 2024). As of December 2023, 151 juvenile manta rays had been identified within the nursery habitat, with 52% being re-sighted and 26% re-sighted over multiple years. New individuals are being identified regularly along southeast Florida (J. Pate unpublished data). In addition, the Flower Garden Banks National Marine Sanctuary and the surrounding banks in the northwest Gulf of America have been described as manta ray nursery habitat (Stewart et al., 2018a). These nursery habitats were described based on frequent observations of immature individuals in these areas, high site fidelity with individuals remaining in the areas, and extended use of these areas by individuals over multiple years (Stewart et al 2018a; Pate and Marshall 2020).

Seasonal Distribution Patterns

In the U.S Atlantic, giant manta rays are distributed from Florida to as far north as New York, with a clear expansion to the north during warmer months. The highest nearshore occurrence is predicted to take place off the Atlantic coast of Florida during April, with the distribution extending northward along the shelf-edge as temperatures warm, leading to higher occurrences north of Cape Hatteras, North Carolina from June to October, and then south of Savannah, Georgia from November to March as temperatures cool (Figure 7; Farmer et al., 2022). These findings are consistent with other lines of evidence that demonstrate that large numbers of giant manta rays are known to migrate to the Atlantic coast of Florida during the spring and summer. Aerial observations of manta ray movements indicate that the Atlantic coast of Florida is potentially an important foraging and reproductive habitat (Pate 2024). Each spring manta rays aggregate off the coast of central and northern Florida, between Indian River County, Florida, and the Florida/Georgia border. Typically, individuals are observed during March of each year in coastal waters off Indian River County, then migrate northward, possibly coinciding with rising water temperatures. Anglers reported that when temperatures range between 68 and 72°F both manta ray and cobia abundance peaked, usually between March and April (Braun et al., 2024), which is consistent with findings in Farmer et al., (2022). Additionally, several lines of evidence indicate that the Mississippi delta region is an important aggregation. Farmer et al. (2022) predicted that the highest nearshore occurrence of giant manta rays in the Gulf of America occurs around the Mississippi River Delta from April to June and again from October to November (Figure 7; Farmer et al., 2022). These findings are supported by directed research and survey efforts, public sightings, and fisheries bycatch data that indicate the Mississippi River Delta is likely an important aggregation site (NMFS, unpublished data).

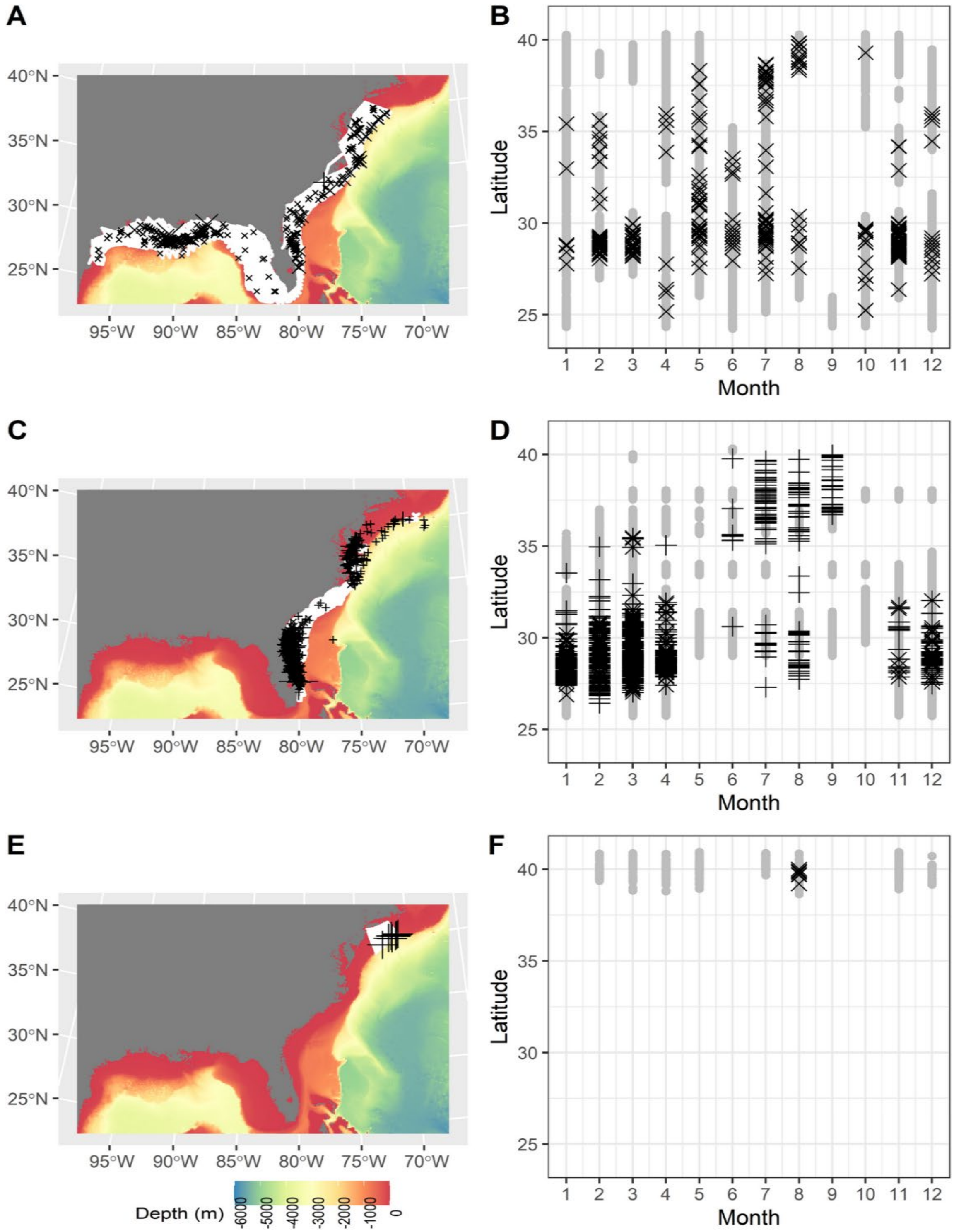

Figure 7. Spatial distribution relative to coarse-scale bathymetry (red = shallow; blue = deep) and survey effort (white lines) and (B) spatio-temporal distribution of survey effort (gray circles) and manta ray sightings (X: on effort, + : off effort; scaled to number reported within survey) by Southeast Fisheries Science Center, (C,D) North Atlantic Right

Whale Consortium, and (E,F) Normandeau Associates aerial surveys for New York State Energy Research and Development Authority. Source: Farmer et al. (2022).

Predators and Competitors

Manta rays are frequently sighted with non-fatal injuries consistent with shark attacks, although the prevalence of these sightings varies by location (Mourier 2012). Deakos et al. (2011) reported that scars from shark predation, mostly on the posterior part of the body or the wing tip, were evident in 24% of reef manta ray individuals (n=70 individuals with injuries) observed at a manta ray aggregation site off Maui, Hawaii. Off eastern Australia, Couturier et al. (2013) observed 23% of reef manta rays had shark scars. Approximately 76% of reef manta rays bear bite wounds of predatory sharks in southern Mozambique (Marshall and Bennett 2010). Because the damage from a shark bite usually occurs in the posterior region of the manta ray, there may be disfigurement leading to difficult clasper insertion during mating or inhibited waste excretion.

Population Dynamics

Population Size

Although the global population size is not known, regional populations have been estimated in Ecuador, Indonesia, Mexico, and Mozambique. Ecuador's Machalilla National Park and the Galapagos Marine Reserve is thought to be home to the largest identified population of giant manta ray in the world, with the estimated population size of 11,022 (95% CI: 9095−13 357) for females and 11,294 (9456−13 490) for males (Hearn et al., 2014; Harty et al., 2022). The next largest population has been noted in Raja Ampat, Indonesia, but is much smaller, estimated at around 1,875 individuals (Beale et al., 2019). The other estimated populations are similar in size, with 1,172 individuals the Revillagigedo Archipelago, Mexico (Cabral et al., 2023), more than 400 individuals in Banderas Bay, Mexico (Domínguez-Sánchez et al., 2023), and 600 individuals Mozambique. Preliminary (uncorrected for availability bias) relative abundance estimates for giant manta rays in the northwest Atlantic Ocean and Gulf of America, U.S., suggest an abundance ranging from approximately 5,000–14,000 individuals with a coefficient of variation between 14–20%, depending on the month (N. Farmer unpubl. data 2023). Preliminary satellite tagging returns from nine individuals suggest manta rays in the southeast spend a median of 14% of their time within depths visible to aerial observers; adjusted estimates for this availability bias suggest 47,802±121,032 (mean ± SD; range 8,206–161,804) individuals in the northwest Atlantic off the eastern U.S. (N. Farmer unpubl. data 2023). Locally, abundance varies substantially and may be based on food availability and the degree that they were, or are currently, being fished. In most regions throughout their range, the number of giant manta rays observed over the years appear to be small (fewer than 1,000 individuals) (NMFS 2024).

Population Variability

The trend of the number of individuals within populations varies widely across the species range, but trends appear stable where they are protected and declining rapidly where fishing pressure is greater (Marshall et al., 2022). For example, sighting trends appear stable where they receive some level of protections, such as Hawaii (Ward-Paige et al., 2013) and Ecuador (Holmberg and Marshall 2018), although individuals sighted in Ecuador seasonally migrate to Peru (A. Marshall unpubl. data 2019) where directed fishing occurs (Heinrichs et al., 2011). Elsewhere, the number of individuals is likely to be declining in places where the species is targeted or caught regularly as bycatch. For example, in southern Mozambique, a 94% decline in diver sighting records occurred over a 15-year period in a well-studied population (Rohner et al., 2017). Similarly, at Cocos Island, Costa Rica, there has been an 89% decline in diver sighting records of giant manta rays over a 21-year period (White et al., 2015). These steep declines have occurred in less than one-generation length (29 years) (Marshall et al., 2022). Although sparse, the available data suggest that target fisheries in some regions have rapidly depleted localized populations of the giant manta ray; local extinction is suspected to have occurred in many parts of their historical range. Globally, the suspected population reduction is 50–79% over three generation lengths, with a further population reduction suspected over the next several generations based on current and ongoing threats and exploitation levels, steep declines in monitored populations, and a reduction in area of occupancy (Marshall et al., 2022).

Range and Distribution

Within the Northern hemisphere, the giant manta ray has been documented as far north as the following locations: New York, U.S. and the Azores Islands in the Atlantic Ocean region; the Sinai Peninsula, Egypt in the Indian Ocean region; and Mutsu Bay, Aomori, Japan and southern California, U.S. in the Pacific Ocean region (Figure 8; Lawson et al., 2017; Kashiwagi et al., 2010; Moore 2012; CITES 2013; Knochel et al., 2022; Farmer et al., 2022). In the Southern Hemisphere, the species has been observed as far south as Peru in the eastern Pacific Ocean, Uruguay and St. Helena Island in the Atlantic Ocean, South Africa and Australia in the Indian Ocean, and off Tasmania, New Zealand, and French Polynesia in the western and central Pacific Ocean (Figure 8; Lawson et al., 2017; Mourier 2012; CITES 2013,; Carpentier et al., 2019).

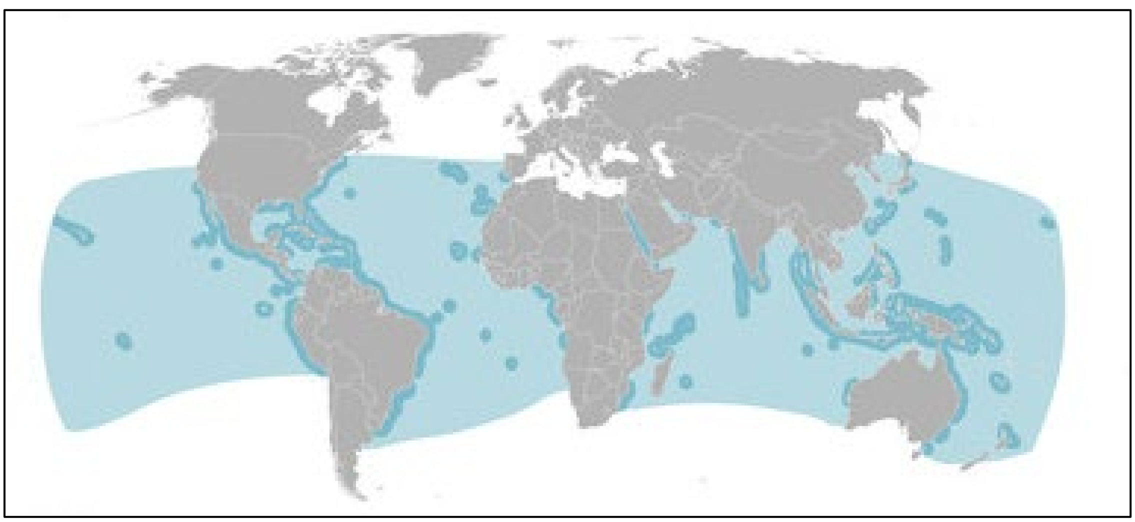

Figure 8. Geographic range of Mobula birostris showing confirmed locations (extend in dark blue) as well as presumed range (possibly extant in light blue) (Lawson et al. 2017).

Biotic and Abiotic Factors Dictating Range / Distribution

Giant manta rays are commonly sighted in aggregations at many locations throughout their range, including: Similan Islands (Thailand); Raja Ampat (Indonesia); Sharm el-Sheikh (Egypt); Fuvahmulah and Addu Atolls (Maldives); northeast North Island (New Zealand), Kona, Hawaii (U.S.); U.S. Atlantic and Gulf of America, Brazil; Cabo Verde; Isla de la Plata (Ecuador); Ogasawara Islands (Japan); Isla Margarita and Puerto la Cruz (Venezuela); northern coast of the Yucatan Peninsula; Isla Holbox; Revillagigedo Islands; and Bahia de Banderas (Mexico) (Notarbartolo-di-Sciara and Hillyer 1989; Clark 2010; Kashiwagi et al., 2010; Stewart et al., 2016a; Hilbourne and Stevens 2019; Pate 2024; Farmer et al., 2022; Knochel et al., 2022; Domínguez-Sánchez et al., 2023; Garzon et al., 2023). The timing of these aggregations varies by region and seem to correspond with the movement of zooplankton, climatic fluctuations (e.g., El Niño Southern Oscillation), current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior (Couturier et al., 2012; Beale et al., 2019; Domínguez-Sánchez et al., 2023, Garzon et al., 2023). For example, in the U.S. Atlantic and Gulf of America, the distribution of manta rays was found to be influenced primarily by sea surface temperature, with a clear expansion to the north during warmer months (Farmer et al., 2022). Additionally, within the preferred thermal range (approximately 68–86°F), manta rays occurred most frequently either nearshore or along the continental shelf-edge, at locations best predicted by proxies for productivity such as thermal fronts, bathymetric slope, and high chlorophyll-a concentration (Farmer et al., 2022).

Threats

Past and Current Threats Resulting in Population Declines

The largest cause of direct mortality of giant manta rays globally is targeted fishing and capture as bycatch (NMFS 2024), though they face a range of other threats including marine pollution, vessel strikes, and entanglement (NMFS 2024). In addition, while there is no direct evidence, there is concern for this species as a result of impacts associated with climate change (Essumang 2010, Ooi et al., 2015, Stewart et al., 2018).

Fisheries (Foreign)

Giant mana rays are both targeted and caught incidentally in industrial and artisanal fisheries (Couturier et al., 2012; Stewart et al., 2018). These rays are captured in a wide range of gear types including harpoons, drift nets, purse seine nets, gill nets, traps, trawls, and longlines. Their coastal and offshore distribution, and tendency to aggregate, makes giant manta rays particularly susceptible to bycatch in purse seine and longline fisheries and targeted capture in artisanal fisheries (Duffy and Griffiths 2017). In particular, giant manta rays are easy to target because of their large size, slow cruising speed, tendency to aggregate, predictable habitat use, and lack of human avoidance (Couturier et al., 2012). Manta ray catches have been recorded in at least 30 large and small-scale fisheries covering 25 countries (Lawson et al., 2016). The largest documented target fisheries are Indonesia, the Philippines, and India (NMFS 2024). While many artisanal fisheries have grown to meet international trade demand for gill plates as discussed below, some still target these manta rays mainly for food and local products (Essumang 2010, Rohner et al., 2017).

Federal Fisheries (Southeast U.S.)

The giant manta ray is caught as bycatch in a number of federally-managed U.S. commercial fisheries operating in the Atlantic Ocean and Gulf of America. Data comes from limited fisheries observer programs. Information was severely lacking on catch and fishing effort of the species in the Atlantic at the time of listing because rays were not identified to species in observer programs. In 2019, fisheries observer programs in the Southeast began identifying and recording bycatch for the giant manta ray, thus providing a better understanding of U.S. commercial fisheries interactions with giant manta rays. Based on all available observer data, the Southeast U.S. commercial fisheries that use trawls, pelagic and bottom longlines, gillnet, and hook and line gears occasionally incidentally capture giant manta rays. Shrimp trawl and pelagic longline gears appear to interact with giant manta rays the most, followed by bottom longline, and gillnet. Dispositions of the giant manta rays are recorded at the vessel (i.e., released alive, discarded dead, or disposition unknown) and the poor physical condition of some animals at upon release indicate that animals do die after these encounters. However, post-release mortality is unknown.

The Southeast Shrimp Trawl Fishery

The southeast U.S. shrimp fishery operates in the EEZ in the Gulf of America and U.S. Atlantic, and observer coverage in this fishery is less than 2%. On April 26, 2021, the NMFS Southeast Regional Office (SERO) issued a Biological Opinion (Opinion) on the implementation of the sea turtle conservation regulations under the ESA for shrimp trawls and the authorization of the southeast U.S. shrimp fisheries in federal waters under the Magnuson-Stevens Act (NMFS 2021). As part of the Opinion, an incidental take statement was issued that anticipated the non-lethal capture of 16,780 giant manta rays over 10 years (averaging 1,678 giant manta rays per year) in the shrimp trawl fishery. No giant manta ray mortalities were anticipated because there were no records of lethal interactions at that time. The incidental take estimate was based on 1 year of data, which included 12 non-lethal interactions documented during that time, and was highly uncertain.

In 2023, NMFS reinitiated this consultation because there were giant manta ray mortalities documented by the observer program. Between 2019 and April 2024, the SEFSC shrimp observer program observed approximately 37 giant manta rays were incidentally captured, with 26 released alive, 4 dead, and 7 discarded with an "unknown" disposition (NMFS unpublished data). The majority of interactions were recorded off the coast of Louisiana, followed by coastal areas near the Florida/Georgia border. The lack of many interactions observed in the U.S Atlantic may be due to very low observer coverage, as several recent studies indicate that north and central Florida is likely an important habitat for giant manta ray migration, foraging and reproduction (Farmer et al., 2022; Pate 2024). The majority of interactions have occurred in Federal waters, although there are records in State waters as well.

The Pelagic Longline Fishery

The Pelagic Longline Fishery for Atlantic Highly Migratory Species comprises relatively distinct segments including: Caribbean, Gulf of America, Florida east coast, South Atlantic bight, mid-

Atlantic bight, northeast coastal Atlantic, northeast distant waters, Sargasso Sea, and Offshore waters. Observer coverage is maintained at a minimum of 8%, but some years have higher coverage (NMFS 2020).

On May 15, 2020, NMFS SERO issued an Opinion on the operation of the Pelagic Longline Fishery for Atlantic Highly Migratory Species in federal waters under the Magnuson-Stevens Act (NMFS 2020). As part of the Opinion, an incidental take statement was issued that anticipated the non-lethal capture of 366 giant manta rays and 6 mortalities over 3 years in the Pelagic Longline Fishery. At that time, the incidental take estimate was uncertain as there was limited incidental capture data of giant manta rays, including mortality data. In addition, uncertainty surrounded species identifications made by the observers as most records pre-dated the ESA listing of giant manta rays and subsequent observer training. In 2022, the NMFS reinitiated the consultation because the number of mortalities documented by the observers have exceeded what was authorized in the 2020 Opinion. From 2020 through 2023, observers (9.9% coverage) recorded 9 giant manta rays captured in pelagic longline gear, with 3 documented at-vessel mortalities. These captures occurred in the mid-Atlantic bight, northeast coastal Atlantic, and Gulf of America fishing zones. Of note, the majority (approximately 71%) of mobulid bycatch records from 2019–2023 lacked identification to the species level; it is unclear what percentage of these records were comprised of giant manta ray.

The Coastal Migratory Pelagic Fishery

The Coastal Migratory Pelagic (CMP) Fishery operates in the Atlantic Ocean and Gulf of America. The fishery primarily targets king mackerel, Spanish mackerel, and the Gulf of America Migratory Group of cobia. The main gear types used in the CMP fishery are hook-and-line (primarily trolling), cast net, and gillnet. Diver-held spear guns are also a main gear type specific to cobia. Interactions with manta rays were observed in 2018 for commercial gillnet sets targeting Spanish mackerel. No interactions with giant manta rays for the gill net component of the king mackerel fishery were reported or observed (NMFS 2023). In addition, recreational anglers targeting cobia along Florida's east coast are known to track giant manta ray migrations for the purpose of targeting cobia (Braun et al., 2024). These anglers will seek out and cast at or near giant manta rays to catch the cobia that are associated with the manta rays (e.g., cobia travel beneath manta rays) (Roberts, 2022). This fishing practice primary occurs off Florida's east coast within the boundaries of the Gulf of America Cobia Migratory Group within the Florida East Coast Zone.

On May 1, 2023, NMFS SERO issued an amendment to Biological Opinion on the operation of the CMP Fishery in federal waters under the Magnuson-Stevens Act (NMFS 2023). As part of the Opinion, an incidental take statement was issued that anticipated the non-lethal capture of 714 giant manta rays and 63 mortalities (including post-release mortality) over 3 years in the CMP Fishery. The incidental take is highly uncertain and is based on discard logbook and observer program data from 2010 to 2020. The discards have percent standard error values over 100 indicating a high level of uncertainty (NMFS 2023). A recent study that was published after the 2023 amendment to the CMP Biological Opinion (Braun et al., 2024), found 86% of cobia anglers interviewed along Florida's east coast reported that they had or their clients (charters) had incidentally hooked giant manta rays while casting at manta rays when fishing for cobia

(Braun et al., 2024). In addition, 93% of anglers reported having observed giant manta rays with hooks and training lines or evidence of vessel strike injuries. Overcrowding and increased vessel activity is also a vessel strike concern as anglers have reported seeing an average maximum of 22 boats (range: 1–50) surrounding a single ray or group of rays at the same time (Braun et al., 2024). The most recent information indicates that this fishing practice results in a potentially significant amount of incidental hooking and an increased risk of vessel strike. It is also possible that vessel overcrowding and incidental hooking are disrupting giant manta rays use of this area which is used during migration and is a potentially important foraging and reproductive habitat for this species (Pate 2024).

Atlantic HMS Fisheries (Excluding the HMS Pelagic Longline Fishery)

Fisheries managed under the Consolidated Atlantic HMS FMP (excluding the pelagic longline fishery) operate within Federal waters of the U.S. EEZ. The gear types authorized include bandit gear, bottom longline, buoy gear, gillnets, green-stick, handline, harpoon, purse seine, rod and reel, and speargun. Giant manta rays have been recorded as catch in shark bottom longline gear, in the research fishery. The shark bottom longline research fishery has 100 percent observer coverage. Between 2008 and 2016, there were 2 giant manta rays reported caught in the shark bottom longline research fishery. In addition, Kroetz et al. (2020) reviewed observer data from 1998 to 2017 and documented giant manta ray interactions in drift gillnets (80%, n=6) targeting sharks. All these interactions occurred in the U.S. Atlantic, with the majority occurring along Florida's east coast, followed by nearshore North Carolina (Kroetz et al., 2020). Of the 6 observed interactions, one resulted in a mortality. As part of the Opinion, an incidental take statement was issued that anticipated the non-lethal capture of 9 giant manta rays over 3 years in the Consolidated Atlantic HMS Fishery.

State Fisheries

Various fishing methods used in state commercial and recreational fisheries, including purse seine, gillnets, trawls, pot fisheries, and vertical lines are all known to incidentally take giant manta rays, but information on these fisheries and their impacts on giant manta rays is sparse (NMFS 2024). Most of the state fisheries data are based on extremely low observer coverage, if any, or giant manta rays were not part of data collection; thus, these data provide insight into gear interactions that could occur but are not indicative of the magnitude of the overall problem of fishery bycatch is state fisheries.

Secondary Threats

The largest cause of direct mortality of giant manta rays globally is targeted fishing and capture as bycatch, they face a range of secondary threats including: recreational fishing; vessel strikes; interactions; entanglement; oil spills; marine pollution; and climate change (NMFS 2024). While these threats are known, the extent to which these impacts may affect individual health and overall population fitness is unclear (Couturier et al., 2012; Stewart et al., 2018).

Recreational Fishing

Recreational fishing from private vessels and fishing piers may interact with giant manta rays. For example, giant manta rays have been observed and reported foul-hooked from boats, fishing piers, jetties, and by recreational anglers fishing for sharks during tournaments. Pate and Marshall (2020) found that 27% of the observed giant manta rays in southeast Florida were foul-hooked or entangled in fishing line, and, of those, 38% interacted with fishing gear more than once. More recent data found that of 152 individual manta rays recorded in southeast Florida, 23.7% had interactions with recreational fishing gear and, of those, 61% had multiple interactions (C. Horn., NMFS, pers comm to J. Pate, 2023). These manta rays were commonly seen in the vicinity of fishing piers and inlet jetties (Pate and Marshall, 2020), and anglers have been observed casting at juvenile manta rays (J.H. Pate unpublished data). NMFS has also documented several manta ray captures by anglers targeting sharks from the shore and from vessels (C. Horn unpubl. data). While some fishing interactions may result in minimal permanent injury to the manta ray, they likely cause considerable stress and possible sub-lethal effects. When fishermen have accidentally hooked manta rays, fight times have been over one hour (J. Pate unpubl. data cited in Pate and Marshall, 2020). Fight time is correlated with physiological stress (i.e., lactate production) in elasmobranchs, with smaller sharks producing more lactate than larger sharks. Fishing line entanglement can have non-lethal effects, including truncated cephalic fins (Deakos et al., 2011), deep lacerations to the body (Pate and Marshall 2020), stress, and impaired feeding or swimming. In addition, amputations and disfigurements, specifically those of the cephalic fin, may reduce feeding efficiency, and the absence of this fin may negatively affect size, growth rate and reproductive success (Marshall and Bennett 2010, Deakos et al., 2011, Couturier et al., 2012, Stewart et al., 2018a). While there have been no manta ray deaths directly attributed to recreational fishing, mortality may be cryptic as manta rays are negatively buoyant, reducing the likelihood of dead animals washing ashore.

Vessel Strike

Vessel strikes are evident in every monitored manta ray population across the globe (Stewart et al., 2018a). Spending considerable time at the surface (e.g., while feeding and basking; (Braun et al., 2015) manta rays are especially susceptible to vessel strikes (McGregor et al., 2019). Vessel strikes are spatially variable and are more likely to occur where vessel density and manta ray aggregation along surface waters is high. For example, off the Ningaloo Coast, vessel strikes were highest during the seasonal aggregation of manta rays, which was attributed to an abundance of zooplankton around the area (McGregor et al., 2019). In French Polynesia, observations of manta rays with sub-lethal injuries caused by fishing gear or boat strikes are more likely near inhabited islands than near uninhabited islands (Carpentier et al., 2019). In addition, in some parts of their range, such as the northwest Atlantic, it is likely that the seasonal contraction of suitable manta ray habitat during the warmer months increases their proximity to busy ports, which could pose a serious threat to the species (Garzon et al., 2020). For example, Garzon et al. (2020) found that the Southeast U.S., followed by Venezuela and The Bahamas, had the largest areas of overlap between predicted core manta ray habitat areas and intense commercial vessel traffic (Figure 3).

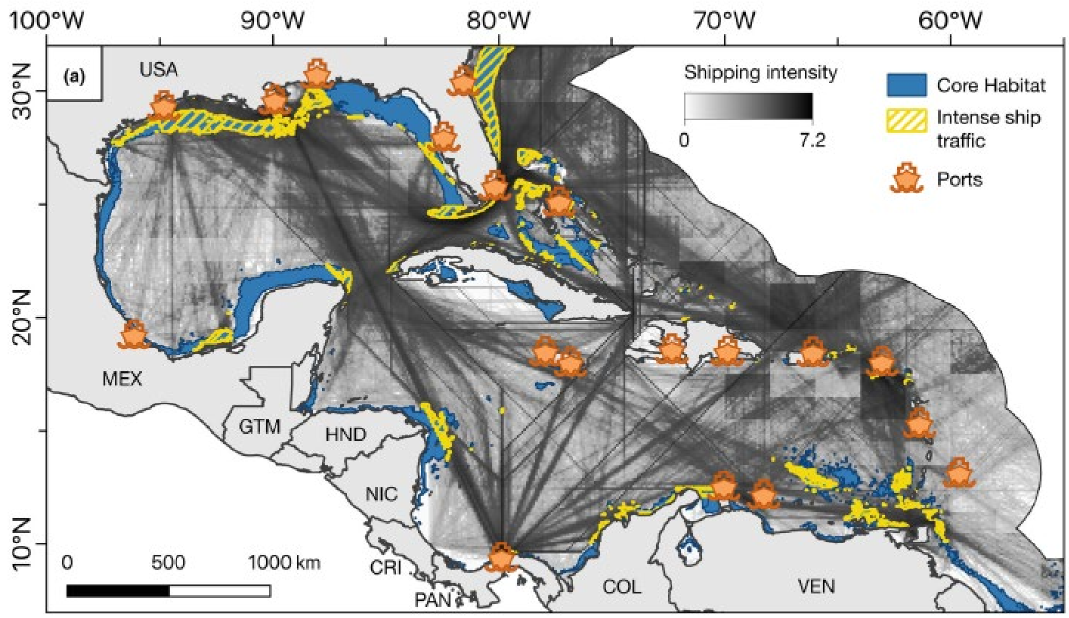

Figure 9. Core manta ray habitat predicted through ensemble ecological niche models with ship traffic, shipping routes and major ports displayed across the western central Atlantic. Source: Garzon et al. (2020)

When comparing the likelihood of vessel strikes on juveniles versus adults, the observed habitat use of juveniles may make them more prone to this threat (Strikes et al., 2022). For example, human activity and vessel traffic is extremely high in the giant manta ray nursery habitat along Florida's Atlantic coast, and vessel strikes are one of the most common sources of injuries to juvenile giant manta rays that frequent the coastal waters there (Pate and Marshall, 2020). Of the known individual giant manta rays (n=179) occurring in southeast Florida, 15 individuals (8.4%) have been recorded with apparently non-lethal vessel strike injuries between June 2016 and December 2023 (J. Pate, pers. comm. to C. Horn, NMFS SERO PRD, January 12, 2024). However, interpreting reported vessel strike data can be difficult due to the fact that many vessel strikes are not detected or go unreported, and there are reporting biases related to spatial coverage of reports (Garzon et al., 2020; McGregor et al., 2019).
One source of underreporting for manta ray vessel strikes is the rapid wound healing characteristic of the species. McGregor et al. (2019) found that manta ray wound healing follows a negative exponential curve, with 37% of the wound closed after 33 days and 95% of the wound closed after 295 days making vessel strike injuries less likely to be identified the longer the wound goes unobserved. Further, researchers note that identifying vessel strike injuries once the wound is mostly closed is exceedingly difficult, especially when most photographs are poor quality making it difficult to observe the injury (C. Horn, NMFS pers comm. to J. Pate, MMF, July 18, 2024).

No information is currently available to estimate mortality rates for giant manta rays from vessel strikes. However, for sea turtles, greater than 75% of vessel strikes result in mortality with only 10-20% of vessel strikes being observed. The mortality rate for giant manta rays is likely higher because the giant manta ray's head and body cavity, which includes all major organs, is large, making up about approximately half of their body's surface area. While surface feeding and basking their body cavity and head are at the surface, unlike sea turtles, the giant manta ray has no hard shell to protect their vital organs, thus, making it more likely that injuries sustained to the head and body will be fatal. It is notable that all the vessel strike injuries documented within the southeast Florida survey domain have occurred on the individual's pectoral fins, not the head or body. Lethal vessel strikes are unlikely to be observed because giant manta rays sink once they die due to their lack of a swim bladder. For these reasons, the giant manta ray mortality rate and unobserved vessel strike death rate may be higher than sea turtles.

Entanglement in Non-fishing lines

The giant manta ray must constantly swim in order to move water over their gills to breathe, and mooring line or rope entanglement can significantly restrict their ability to swim, rapidly leading to asphyxiation and death (Manta Trust 2019a). Entanglement in mooring, anchor line, ropes, and buoy lines can also cause disfigurements and amputations (i.e., missing cephalic lobes) (Braun et al., 2015; Convention on Migratory Species 2014; Couturier et al., 2012; Deakos et al., 2011; Germanov and Marshall 2014; Heinrichs et al., 2011). The eyes on a giant manta ray are positioned on the sides of their heads, limiting their vision in front of them. This makes it less likely that they will see a line directly in front of them as they swim forward. It is likely that giant manta rays become entangled because, when the line makes contact with the front of the head between the cephalic lobes, the animal's reflex response is to close the cephalic lobes, thereby trapping the rope, which entangles the animal as it begins to roll in an attempt to free itself (A. Marshall pers comm to C. Horn, NMFS, 2019). In 2017, a giant manta ray was documented as dead, entangled in a vessel exclusion line (steel cable) near Pompano Beach, Florida. The female measured 2.48 m in disc width and had no other signs of injury or fishing line entanglement. It is likely that the manta ray became entangled in the line and drowned (Pate et al., 2020). In Hawaii, numerous manta rays have been reported to have died or have evidence of injury (i.e., amputations or disfigurements) as a result of entanglement in mooring lines (Deakos, 2011). Mooring line entanglements have resulted in the death of numerous manta rays in the Maldives (Manta Trust, 2019b). There have been several reported incidences of giant manta ray entanglements on vertical lines deployed during oil and gas activities. Oil and gas activities can deploy numerous vertical lines during operation including diver downlines, acoustic buoy release lines, acoustic pinger lanyards, nodal tether cables, and nodal lanyards etc. Similar to mooring line entanglements discussed above, the giant manta ray cannot see a vertical line directly in front of them and they become entangled once the line makes contact with their head. There have been several confirmed reports of giant manta rays becoming entangled in vertical lines deployed by commercial oil and gas divers in the Gulf of America in recent years (C. Horn and N. Famer unpublished data 2022). For example, in 2013, 2021, and 2022, there were reports of giant manta rays entangled in a vertical downlines deployed by oil and gas divers. In addition, commercial oil and gas divers have reported numerous incidences of large rays, possibly giant manta rays in close proximity to underwater operations. It is thought that zooplankton are attracted to the underwater lights deployed by commercial divers. The amassing of zooplankton may be attracting giant manta rays to underwater operation sites where vertical lines are deployed increasing the risk of entanglement (C. Horn, NMFS, personal observation).

Marine Pollution

In locations with high densities of plastic debris and microplastics, giant manta rays may directly ingest plastic marine debris and microplastics, the consequences of which may include exposure to toxic plastic additives and persistent organic pollutants (POPs) (Stewart et al., 2018; Germanov et al., 2019). Studies have found elevated levels of some heavy metals in ray tissues (Essumang 2010; Ooi et al., 2015) and low levels POPs (Germanov et al., 2019). Phthalates or POPs have been recorded in tissue samples of baleen whales, basking sharks, whale sharks, and manta rays in areas with high levels of microplastic pollution (Fossi et al., 2014, 2016, 2017; Germanov et al., 2019), indicating that filter feeding organisms are likely bioaccumulating these pollutants as a result of plastic ingestion. A number of studies have demonstrated that ingesting indigestible particles (e.g., microplastics) can block adequate nutrient absorption and cause damage to the digestive tract. Microplastics can also have high levels of toxins and POPs, and introduce these toxins to the animals via ingestion (Jakimska et al., 2011; Germanov et al., 2018). These toxins can bioaccumulation over decades, in long-lived filer feeders like the giant manta rays, leading to potentially disruption of biological processes (i.e., endocrine disruption) and potentially altering reproductive fitness (Rochman et al., 2014). Furthermore toxins can be transferred from mother to offspring, potentially impacting growth survival and reproduction of progeny (Galloway and Lewis, 2016; Germanov et al., 2018). Additionally, zooplankton can be contaminated with pollutants and toxins (Fossi et al., 2014) through ingesting microplastics and nanoplastics (Cole et al., 2013; Setälä et al., 2014). This suggests that giant manta rays may be secondary consumers of microplastics and associated pollutants even if they are foraging in locations (or at depths) that do not have high densities of floating microplastics Yet, the implications of exposure to pollution and contaminants on the giant manta ray, remain speculative, especially at the level of individual fitness and population viability (Stewart et al., 2018).

Oil and Gas Activities

A recent ecological vulnerability assessment of elasmobranchs and other large pelagic fish found that giant manta rays have the highest ecological vulnerability (compared to other elasmobranch species) to oil spills within the Gulf of America (Romo-Curiel et al., 2022). Giant manta rays are highly susceptible to the negative health effects associated with oil exposure because they are filter feeders that form seasonal aggregations near oil and gas infrastructure in the Gulf of America (Couturier et al., 2012; Germanov et al., 2018; Stewart et al., 2018). Giant manta rays can also encounter oil while swimming, resting, basking, and feeding in their habitats impacted by oil spills. Because giant manta rays are negatively buoyant, carcasses will sink, suggesting a low probability of observing lethal oil impacts to the species. Direct contact with oil can result in significant health effects and jeopardize the giant manta ray's survival. The effects of ingesting indigestible particles, such as oil, include blocking adequate nutrient absorption and causing mechanical damage to the digestive tract. The giant manta ray may also ingest toxins through their prey since zooplankton can be contaminated with oil and other toxins (Fossi et al., 2014; Cole et al., 2013; Setälä et al., 2014) causing significant health implications. While studies have suggested that highly mobile fish, like the giant manta ray, may actively shift habitat to avoid oil, thereby reducing or minimizing exposure, this active avoidance may alter important behavior (i.e., foraging and reproduction) and migration patterns, resulting in detrimental effects on populations (Romo-Curiel et al., 2022).

Climate Change

Warming oceans cause changes in ocean acidity, oxygen content, oceanic circulation, and primary productivity dynamics, ultimately affecting food web structure and the distribution and availability of manta ray prey. The major impact of climate change on manta rays is likely to be the projected decline in zooplankton biomass in tropical waters (Stewart et al., 2018a). Biogeochemical models predict that between 1980 and 2100 global zooplankton biomass will have declined by 7-16% (Stock et al., 2014), but some regions, particularly those in the tropics, could experience >50% decline in zooplankton biomass (Stock et al., 2014). While it is unknown how this broad-scale decline in zooplankton biomass at the tropics could impact local areas where giant manta rays feed, the most likely outcome is that there will be lower zooplankton biomass available for manta rays. In addition, changes in climate and oceanographic conditions, such as acidification, are also known to affect zooplankton structure (size, composition, and diversity), phenology, and distribution (Guinder and Molinero 2013). As such, the migration paths and locations of both resident and seasonal aggregations of manta rays, which depend on these animals for food, may similarly be altered (Couturier et al., 2012). To date, warming in northern latitudes off the US East Coast appears to have resulted in a significant northerly shift of manta ray distribution (Farmer et al., 2022). Resulting changes in zooplankton availability could affect the behavior and health of giant manta ray populations. For example, shifts in seasonal migration patterns to feeding grounds and nursery areas could have profound impacts on the species' survival. Additionally, some giant manta rays use coral reefs as cleaning stations where small fish remove parasites and dead or diseased skin from their bodies. As sensitive reef habitats degrade due to climate-driven changes, the abundance of cleaning stations and cleaner fish may be reduced (Jones et al., 2004; Graham et al., 2008). The loss of cleaning opportunities can hinder the giant manta ray's ability to reduce parasitic loads and dead tissue, leading to increased disease and reduced survival.

Aquarium Trade

The giant manta ray is traded internationally for display in public aquariums around the globe. Yet, there is limited information available on the number of animals taken from wild populations for the aquarium trade. Several known aquariums display manta rays collected from wild populations for public display. These aquariums include the Georgia Aquarium (United States), Okinawa Churaumi Aquarium (Japan), Nausicaá National Sea Center (France), Atlantis Resort (The Bahamas), S.E.A Aquarium (Singapore), and uShaka Marine World (Durban, South Africa). The available information indicates that some manta rays are transferred among aquariums. For example, the manta ray at UShaka Marine World outgrew its tank, and was eventually transferred to the Georgia Aquarium. While most wild collected individuals remain in captivity, the Atlantis Resort is one facility that has successfully returned 13 individuals to the wild populations. There is limited information available on the total number of individuals collected for exhibition/aquarium purposes and whether those individuals are giant manta rays or reef manta rays. The only international trade data available comes from the CITES Trade Database. Since the giant manta ray was listed under Appendix II in 2016, the CITES Trade Database (https://trade.cites.org/) reports that two giant manta ray export permits were issued, both in 2018, for France to receive two giant manta rays from the United States for exhibition purposes.

With respect to domestic trade, Florida is the only state within the U.S. that authorizes giant manta ray collection for aquarium and exhibition purposes. The Florida Fish and Wildlife Conservation Commission (FWC) authorizes the collection of giant manta ray from the wild under a Special Activity License (SAL) for exhibition purposes. In 2009 and 2010, three giant manta rays were removed from Florida's waters for exhibition purposes for the Georgia Aquarium. More recently, from 2019 to 2022, the FWC has issued 17 SALs for the collection of giant manta rays for exhibition purposes. These SALs were issued to a number of aquarium facilities that were not previously known to exhibit/display manta rays, including: Nausicaá National Sea Center (France), Hainan Ocean Paradise (Hainan, China), Rizhao Ocean Park (Shandog, China), Changxing Taihu Longzhimeng Sea World (Shanghai, China), Chongqing Andover Ocean Park (Chongqing, China), SeaWorld Abu Dhabi (United Arab Emirates), and The National Aquarium LLC (Maryland, United States) (L. Gregg pers comm to C.Horn July 18, 2023). Despite the receiving the SALs, these facilities were not successful in collecting any individuals from Florida waters. In addition, no CITES export permits were issued for the collection licensed by the FWC. The FWC sets its annual collection quota based on the traditional level of collection request that the state has received for exhibition purposes (L. Gregg pers com to C. Horn, July 18, 2023).

5 ENVIRONMENTAL BASELINE

5.1 Overview

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

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

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

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

The status of each species in the action area that are likely to be adversely affected by the proposed action, as well as the threats to these species, is supported by the species accounts in Section 4 (Status of the Species).

As stated in Section 2.2 (Action Area), the proposed action occurs in Grand Isle, Jefferson Parish, Louisiana. The project site is located within the Grand Isle State Park along the northeastern shoreline of Grand Isle immediately adjacent to the Gulf of America. For the purposes of this federal action, the action area includes the nearshore areas in which construction will take place. Around the fishing pier demolition and replacement pier construction, the action area extends within a 1,120.3 ft (341.47 m) buffer, which encompasses the area where behavioral effects to sea turtles and the giant manta ray related to noise from pile driving may occur, as well as the general area accessible by recreational anglers fishing from the pier (i.e., casting distance or approximately 200 ft). The total acreage within this defined action area is approximately 88.7 ac. Substrate within the action area consists primarily of sand and silt soils with water depths ranging between 0 ft and 10 ft. There are no corals, mangroves, SAV, or designated critical habitat within the proposed action area.

5.2.1 Sea Turtles

There have been no reported recreational hook-and-line captures of ESA-listed sea turtles at the Grand Isle State Park Fishing Pier according to the STSSN data for the years 2007-2016. However, based on the best available species life history data and the STSSN recreational hook-and-line capture and entanglement data (Table 2), green (North Atlantic DPS), Kemp's ridley, loggerhead (Northwest Atlantic DPS) sea turtles may be located in the action area and adversely affected by the recreational hook-and-line fishing that will occur at the Grand Isle State Park Fishing Pier upon completion of the proposed action. All of these species are migratory, traveling for foraging or reproduction purposes. The inshore waters around Grand Isle may be used by these sea turtles as developmental and foraging habitat. NMFS believes that no individual sea turtle is likely to be a permanent resident of the inshore waters of the action area, although some individuals may be present at any given time. These same individuals will migrate into offshore waters, as well as other areas of the Gulf of America, Caribbean Sea, and North Atlantic Ocean at certain times of the year, and thus may be impacted by activities occurring in these areas. The status of this species in the action area, as well as the threats to this species, is supported by the species accounts in Section 4 (Status of the Species).

5.2.2 Giant Manta Ray

NMFS is not aware of any reported recreational hook-and-line captures of a giant manta ray at the Grand Isle State Park Fishing Pier. Giant manta rays can occur in coastal bays, intracoastal waterways, tidal inlets, and in estuarine systems (e.g., sounds and lagoons). Giant manta rays are observed feeding in tidal outflows, inlets, and river mouths (feeding around outfall plumes) (Adams and Amesbury 1998; Milessi and Oddone 2003; Pate and Marshall 2020; Farmer et al. unpublished). They are also commonly observed swimming near or underneath public fishing piers where they may become foul-hooked. Due to the pier's position in relative close proximity to an inlet, we believe giant manta ray may be adversely affected by recreational fishing that will occur at the pier upon completion of the proposed action. NMFS believes that no individual giant manta ray is likely to be a permanent resident of the action area, although some individuals may be present at any given time. These same individuals will migrate into coastal and offshore waters of the Gulf of America and the North Atlantic Ocean, and thus may be affected by activities occurring there. Therefore, the status of giant manta ray in the action area, including the threats, are the same as those discussed in Section 4.3.

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

5.3.1 Federal Actions

ESA Section 7 Consultations

We have consulted on several projects within the greater Barataria Bay area where the project is located. However, other than the proposed action, which is a reinitiation of a previous Opinion (SERO-2018-00147; issued October 23, 2019), no other federally permitted projects are known to have occurred within the action area, as per a review of the NMFS SERO PRD's completed consultation database by the consulting biologist on January 3, 2025. Other fishing piers (outside of the action area) that also require federal permits have been subject to formal consultation, resulting in Biological Opinions and measures to minimize the impact of associated take. Those consultations generally found fishing piers adversely affect sea turtles and giant manta rays via incidental hooking and entanglement by actively-fished lines; discarded, remnant, or broken-off fishing lines; and other debris.

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 NOAA Fisheries Authorizations and Permits for Protected Species (APPS) database (https://apps.nmfs.noaa.gov/) by the consulting biologist on January 3, 2025, there were 6 active scientific research permits applicable to green, Kemp's ridley, and loggerhead sea turtles within the State of Louisiana, all of which may occur in the action area. These permits allow the capture, handling, sampling, and release of these turtle species (all life stages including hatchlings) with the purpose of gaining better scientific knowledge.

5.3.2 State and Private Actions

<u>Recreational Fishing</u>

Recreational fishing as regulated by the State of Louisiana can affect green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray or their habitats within the action area. Pressure from recreational fishing in and adjacent to the action area is likely to continue. As stated above, there have been no reported recreational hook-and-line captures of ESA-listed sea turtles at the Grand Isle State Park Fishing Pier according to the STSSN data for the years 2007-2016. We have no way of knowing how many unreported captures of these species may have occurred at the pier in the past. Observations of state recreational fisheries have shown that 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 (1998; 2000).

Giant manta ray is incidentally captured by recreational fishers using vertical line (i.e., handline, bandit gear, and rod-and-reel). Researchers frequently report giant manta rays having evidence of recreational gear interactions along the east coast of Florida (i.e., manta rays have embedded fishing hooks with attached trailing fishing line) (J. Pate, Florida Manta Project, unpublished data). Internet searches also document recreational interactions with giant manta rays. For example, recreational fishers will search for giant manta rays while targeting cobia, as cobia often accompany giant manta rays. Giant manta rays are commonly observed swimming near or underneath public fishing piers where they may become foul-hooked.

5.3.3 Marine Debris, Pollution, and Environmental Contamination

Sources of pollutants along the coastal areas that may affect green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray include atmospheric loading of 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). In addition, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, and boat traffic can degrade marine habitats used by sea turtles (Colburn et al. 1996). 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. Additionally, anthropogenic marine debris is known to affect ESA-listed sea turtle species and giant manta ray; however, the effects are difficult to measure. Where possible, conservation measures are being implemented to monitor or study the effects to sea turtles from these sources.

The development of marinas and docks in inshore waters can negatively affect 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.

5.3.4 Acoustic Impacts

Acoustic effects on ESA-listed sea turtles and giant manta ray are a known impact to these species and they are difficult to measure. Where possible, conservation actions are being implemented to monitor or study the effects to sea turtles from these sources.

5.3.5 Stochastic Events

Seasonal stochastic (i.e., random) events, such as hurricanes or cold snaps, occur in Louisiana and can affect green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray in the action area. These events are unpredictable and their effect on the recovery of these ESA-listed species is unknown; yet, they have the potential to directly impede recovery if animals die as a result or indirectly if important habitats are damaged.

5.3.6 Climate Change

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

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

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

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

6 EFFECTS OF THE ACTION

6.1 Overview

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

In this section of our Opinion, we assess the effects of the action on listed species that are likely to be adversely affected. The analysis in this section forms the foundation for our jeopardy analysis in Section 8. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Where data are limited or equivocal, we have occasionally needed to make reasonable determinations based upon our professional judgment to bridge the gap in the available data. 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 all instances, the approach to our analysis is explained, including how uncertainty, causation, and the choice among a range of values are evaluated and addressed.

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

6.2.1 Routes of Effect That Are Not Likely to Adversely Affect ESA-Listed Species Considered for Further Analysis

Green (North Atlantic DPS), Kemp’s ridley, and loggerhead (Northwest Atlantic DPS) sea turtles, and giant manta ray may be physically injured if struck by construction equipment, support vessels, or materials. However, we believe this route of effect is extremely unlikely to occur because mobile species, such as these, are able to avoid slow-moving equipment, support vessels, and the placement of materials. The applicant's implementation of our *Protected Species Construction Conditions* (NMFS 2021) and *Vessel Strike Avoidance Measures* (NMFS 2021) will further reduce the risk to these species. Operation of moving equipment shall cease immediately if a protected species is seen within a 150-ft radius of operations. Activities may not resume until the protected species has departed the project area of its own volition or 20 minutes have passed since the animal was last seen in the area. Further, construction would be limited to daylight hours so construction workers are able to see protected species, if present, and avoid interactions with them.

The action area contains shallow-water habitat that may be used by green, Kemp's ridley, and loggerhead sea turtles, and giant manta ray for foraging and refuge. These species may be affected if they are temporarily unable to use the site for forage or refuge habitat due to avoidance of construction activities and related noise. These effects will be insignificant given the project's limited footprint and availability of similar habitat nearby. Given the action area's lack of seagrass, use of the in-water area by sea turtle species for foraging and refuge is expected to be infrequent. Any disturbances to species would be temporary, limited to approximately eleven months of in-water construction during daylight hours only, after which animals will be able to return to the site. Additionally, the new pier will feature sea turtle friendly lighting on the pier and beneath the pier. Specifically, the lighting beneath the pier will be installed with a cover and pointed toward the shore to reduce Gulf-facing lighting that could attract sea turtles to the pier.

The NMFS educational signs *"Save Dolphins, Sea Turtles, Sawfish, and Manta Ray"* and *"Do Not Catch or Harass Sea Turtles"* will be installed in a visible location upon completion of the proposed action prior to the fishing pier being open to the public. We believe the placement of these educational signs will reduce the likelihood of recreational hook-and-line interactions with ESA-listed species in the action area. The signs will provide information to the public on how to avoid and minimize encounters with these species as well as proper handling techniques. The signs will also encourage anglers to report sightings and interactions, thus providing valuable distribution and abundance data to researchers and resource managers. Accurate distribution and abundance data allows management to evaluate the status of the species and refine conservation and recovery measures.

Green, Kemp's ridley, and loggerhead sea turtles, and giant manta ray may be injured due to entanglement in improperly discarded fishing gear resulting from future use of the fishing pier after completion of the proposed action. We believe this route of effect is extremely unlikely to occur. To the best of our knowledge, there has never been a reported entanglement with any of these species at this fishing pier. To help further reduce the risk of entanglement in improperly discarded fishing gear, monofilament recycling bins and trash receptacles will be placed on the fishing pier, will be clearly marked, and will be maintained/emptied regularly to reduce the probability of trash and debris entering the water. We expect that anglers will appropriately dispose of fishing gear when disposal bins are available. Additionally, annual underwater fishing debris cleanup around the pier will be conducted to remove any fishing line, nets, and other debris and trash from the water, minimizing the accumulation of fishing line over time.

Noise created by pile driving activities can physically injure animals or change animal behavior in the affected areas. Animals can be physically injured in two ways. First, immediate adverse effects can occur if a single noise event exceeds the threshold for direct physical injury. Second, adverse physical effects can result from prolonged exposure to noise levels that exceed the daily cumulative sound exposure level for the animals. Noise can also interfere with an animal's behavior, such as migrating, feeding, resting, or reproducing and such disturbances could constitute adverse behavioral effects.

When an impact hammer strikes a pile, a pulse is created that propagates through the pile and radiates sound into the water, the ground substrate, and the air. Pulsed sounds underwater are typically high volume events that have the potential to cause hearing injury. In terms of acoustics, the sound pressure wave is described by the peak sound pressure level (PK, which is the greatest value of the sound signal), the root-mean-square pressure level (RMS, which is the average intensity of the sound signal over time), and the sound exposure level (SEL, which is a measure of the energy that takes into account both received level and duration of exposure). Further, the cumulative sound exposure level (SELcum) is a measure of the energy that takes into account the received sound pressure level over a 24-hour period. Please see the following website for more information related to measuring underwater sound and the NMFS-accepted pile driving sound measurement thresholds for species in the NMFS Southeast Region: https://www.fisheries.noaa.gov/southeast/consultations/section-7-consultation-guidance.

We use the proposed action's pile driving information and NMFS Multi-Species Pile Driving Tool (dated May 2022) to calculate the radii of physical injury and behavioral effects on ESA-listed species that may be located in the action area based on the NMFS-accepted pile driving sound measurement thresholds for species in the NMFS Southeast Region reference above.

The project proposal includes impact pile driving of up to six piles per day during daylight hours only. Pile driving will occur in an open-water environment. We define an open-water environment as any area where an animal would be able to move away from the noise source without being forced to pass through the radius of noise effects. Based on our noise calculations, the installation of up to six 16-in diameter concrete piles by impact hammer will have a single-strike or PK injurious noise effects to a radius of 1.1 ft (0.3 m) for ESA-listed fishes. We believe PK injurious noise effects are extremely unlikely to occur because this distance is within the 150-ft (46-m) "stop-work" radius defined in NMFS SERO's *Protected Species Construction Conditions* (revised 2021). Additionally, the SELcum of multiple pile strikes over the course of a day may cause injury to ESA-listed fish and sea turtles at a radius of up to 241.4 ft (73.6 m). Due to the mobility of sea turtles and ESA-listed fish species, and because the project occurs in open water, we expect them to move away from noise disturbances. Because we anticipate the animal will move away, we believe that an animal's suffering physical injury from noise is extremely unlikely to occur. Construction personnel will cease construction activities if an ESA-listed species is sighted per NMFS SERO's Protected Species Construction Conditions. Thus, we believe the likelihood of any injurious SELcum effects is extremely unlikely to occur. An animal's movement away from the injurious impact zone is a behavioral response, with the same effects discussed below.

The installation of concrete piles using an impact hammer could result in behavioral effects at a radius of 1,120.3 ft (341.5 m) for ESA-listed fish species and 24.1 ft (7.4 m) for ESA-listed sea turtles. The radius for behavioral effects for sea turtles is within the 150-ft (46-m) "stop-work" radius defined in SERO's *Protected Species Construction Conditions* (revised 2021).Therefore, we anticipate any behavioral effects to these species will be insignificant. Additionally, due to the mobility of ESA-listed fish species and sea turtles, we expect them to move away from noise disturbances in these open-water environments. Because there is similar habitat nearby, we believe behavioral effects will be insignificant. If an individual chooses to remain within the behavioral response zone, it could be exposed to behavioral noise impacts during pile installation. Since installation will occur only during the day, these species will be able to resume normal activities during quiet periods between pile installations and at night. Therefore, we anticipate any behavioral effects will be insignificant.

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

As stated above, we believe hook-and-line gear commonly used by recreational anglers fishing at the Grand Isle State Park Fishing Pier may adversely affect green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray. In Section 6.2.2.1-6.2.2.3, we provide more detail on the potential effects of entanglement in actively fishing recreational gear, hooking by recreational fishing gear, and trailing line to these species from hook-and-line gear. Section 6.4 addresses how we estimate future captures of sea turtles. Section 6.5 addresses how we estimate future captures of giant manta ray.

6.2.2.1 Entanglement in Actively-Fished Gear

Sea turtles are particularly prone to entanglement as a result of their body configuration and behavior. Records of stranded or entangled sea turtles reveal that hook-and-line gear can wrap around the neck, flipper, or body of a sea turtle and severely restrict swimming or feeding. If the sea turtle is entangled when young, the fishing line becomes tighter and more constricting as the sea turtle grows, cutting off blood flow and causing deep gashes, some severe enough to remove an appendage. Sea turtles have been found entangled in many different types of hook-and-line gear. Entangling gear can interfere with a sea turtle's ability to swim or impair its feeding, breeding, or migration. Entanglement may even prevent surfacing and cause drowning.

Fishing line entanglement can cause effects to giant manta ray, including injury to cephalic fins (Deakos et al. 2011), stress, deep lacerations to the body, and impaired feeding or swimming. The effects from entanglement are considered sub-lethal to giant manta ray because they do not immediately result in death, with documented evidence that manta rays can recover and survive post-injury (Pate and Marshall 2020).

6.2.2.2. Hooking

Sea turtles are also injured and killed by being hooked. Hooking can occur as a result of a variety of scenarios, some depending on the foraging strategies and diving and swimming behavior of the various species of sea turtles. Sea turtles are either hooked externally in the flippers, head, shoulders, armpits, or beak (i.e., "foul-hooking", when an animal is hooked anywhere on the body without having taken the bait in its mouth), or internally inside the mouth or when the animal has swallowed the bait (Balazs et al. 1995). Swallowed hooks are of the greatest threat. A sea turtle's esophagus (throat) is lined with strong conical papillae directed towards the stomach (White 1994). The presence of these papillae in combination with an S-shaped bend in the esophagus make it difficult to see hooks when looking through a sea turtle's mouth, especially if the hooks have been deeply ingested. Because of a sea turtle's digestive structure, deeply ingested hooks are also very difficult to remove without seriously injuring the turtle. A sea turtle's esophagus is also firmly attached to underlying tissue; thus, if a sea turtle swallows a hook and tries to free itself or is hauled on board a vessel, the hook can pierce the sea turtle's esophagus or stomach and can pull organs from its connective tissue. These injuries can cause the sea turtle to bleed internally or can result in infections, both of which can kill the sea turtle. If an ingested hook does not lodge into, or pierce, a sea turtle's digestive organs, it can pass through the digestive system entirely (Aguilar et al. 1995; Balazs et al. 1995) with little damage (Work 2000). For example, a study of loggerheads deeply hooked by the Spanish Mediterranean pelagic longline fleet found ingested hooks could be expelled after 53 to 285 days (average 118 days) (Aguilar et al. 1995). If a hook passes through a sea turtle's digestive tract without getting lodged, the hook probably has not harmed the turtle.

Hook-and-line gear commonly used by recreational anglers fishing from fishing piers can adversely affect giant manta ray via hooking or foul-hooking. The effects from hooking and foul-hooking are considered sub-lethal to giant manta ray because they do not immediately result in death, with documented evidence that manta rays can recover and survive post-injury (Pate and Marshall 2020).

6.2.2.3 Trailing Line

Trailing line (i.e., line left on an animal after it has been captured and released) poses a serious risk to sea turtles. Line trailing from a swallowed hook is also likely to be swallowed, which may irritate the lining of the digestive system. The line may cause the intestine to twist upon itself until it twists closed, creating a blockage ("torsion"), or may cause a part of the intestine to slide into another part of intestine like a telescopic rod ("intussusception") which also leads to blockage. In both cases, death is a likely outcome (Watson et al. 2005). The line may also prevent or hamper foraging, eventually leading to death. Trailing line may also become snagged on a floating or fixed object, further entangling a turtle and potentially slicing its appendages and affecting its ability to swim, feed, avoid predators, or reproduce. Sea turtles have been found trailing gear that has been snagged on the sea floor, or has the potential to snag, thus anchoring them in place (Balazs 1995). Long lengths of trailing gear are more likely to entangle the sea turtle, eventually leading to impaired movement, constriction wounds, and potentially death.

The effects to giant manta ray from trailing line are the same as those discussed above under Entanglement in Actively Fished Gear.

6.3 Estimating Hook-and-Line Interactions with Sea Turtles

6.3.1 Estimating Future Reported Hook-and-Line Interactions with Sea Turtles

We believe the best available data to estimate future reported recreational hook-and-line interactions with sea turtles at public fishing structures comes from the historic reported captures at similar structures obtained from STSSN data, and any additional information regarding captures at the structure under consultation. We believe that using this dataset is the most accurate representation of the likely range of future interactions in the action area given the rarity of expected interactions and variability in species presence and angler behavior. The STSSN data contains number and location of sea turtle recreational hook-and-line captures that were reported to the STSSN; it does not provide the total number of potential public fishing structures available in a particular zone, and NMFS does not have that information. Below, we provide additional discussion regarding why this is the best available information to estimate the expected annual number of reported recreational hook-and-line captures of sea turtles at the Grand Isle State Park Fishing Pier in the future.

As previously stated, the Grand Isle State Park Fishing Pier is located in the Gulf of America-facing waters of Zone 12. The STSSN dataset contains no reported captures of sea turtles at the Grand Isle State Park Fishing Pier (recreational hook-and-line or otherwise; years 2007-2016). However, there is a combined total of 717 reported recreational hook-and-line captures of sea turtles at 23 similar inshore, public fishing structures in Zone 12 during this period. Because these 23 fishing structures are in a similar habitat and location as the Grand Isle State Park Fishing Pier (i.e., Zone 12), we assume sea turtle behavior, density, and species composition are comparable at all locations. Because the fishing structures are of a similar size, they likely have comparable angler effort. Further, we assume anglers fishing from both of these structures use similar baits, equipment, and fishing techniques. Therefore, even though the historic reported hook-and-line captures are different between these structures, the potential for interactions with sea turtles is likely comparable at all 23 locations within Zone 12.

Whether interactions with sea turtles are reported varies depending on a number of factors, including whether there are educational signs encouraging reporting and angler behavior; sometimes anglers do not report encounters with ESA-listed species due to concerns over their personal liability or public perception at the time of the capture even if there are posted signs. Given this variability, it is difficult to estimate reporting behavior. However, we assume that similar fishing structures within the same statistical fishing zone (in this case, Zone 12) would have similar reporting rates. Because piers in the same reporting zone are in similar geographic locations, we assume public perception about reporting and angler reporting behavior is likely the same. Therefore, even though the historic reported hook-and-line captures are different between these structures, the potential for reported captures is the same at both locations.

Thus, we believe the best available data to estimate the number of future reported recreational hook-and-line captures of sea turtles at the Grand Isle State Park Fishing Pier is the average of the historic reported recreational hook-and-line captures at the similar fishing structures in the Zone 12 STSSN dataset and the absence of reported captures at the Grand Isle State Park Fishing Pier. Averaging the Zone 12 data helps smooth variability in both the potential for interactions (i.e., number and species composition) and in reporting behavior among the locations and over time, providing for a more accurate overall estimate of future reported captures at the consultation pier. There is no additional information that can be used to estimate potential reported interactions.

To calculate the average number of reported hook-and-line captures at these similar fishing structures in Zone 12, we use the available STSSN data and following equation:

$$Average\ Reported\ Captures\ Per\ Stucture\ in\ 10\ years$$
$$= Sum\ of\ Reported\ Captures\ in\ 10\ years \div 23\ Locations$$
$$= (717) \div 23$$
$$= 31.1174\ per\ structure, over\ 10\ years$$

To calculate the estimated expected annual number of reported recreational hook-and-line captures of sea turtles at the Grand Isle State Park Fishing Pier, we refer to the information above and use the following equation:

$$Expected\ Annual\ Reported\ Captures$$
$$= Average\ Reported\ Captures\ Per\ Structure\ in\ 10\ years \div 10\ years$$
$$= 31.1174 \div 10$$
$$= 3.1117\ reported\ captures\ per\ year\ (Table\ 5, Line\ 1)$$

6.3.2 Unreported Hook-and-Line Interactions with Sea Turtles

While we believe the best available information for estimating future captures at the Grand Isle State Park Fishing Pier are the reported captures at public fishing piers in Zone 12, we also recognize the need to account for unreported captures. In the following section, we estimate the number of unreported recreational hook-and-line-captures at the pier. To the best of our knowledge, only two fishing pier surveys aimed at collecting data regarding unreported recreational hook-and-line captures of ESA-listed species have been conducted in the Southeast. One is from Charlotte Harbor, Florida, and the other is from Mississippi.

The fishing pier survey in Charlotte Harbor, Florida, was conducted at 26 fishing piers in smalltooth sawfish critical habitat (Hill 2013). During the survey, 93 anglers were asked a series of open-ended questions regarding captures of sea turtles, smalltooth sawfish, and dolphins, including whether they knew these encounters were required to be reported and if they did report encounters. The interviewer also noted conditions about the pier including if educational signs regarding reporting of hook-and-line captures were present at the pier. Hill (2013) found that only 8% of anglers would have reported a sea turtle hook-and-line capture (i.e., 92% of anglers would not have reported a sea turtle capture).

NMFS conducted the fishing pier survey in Mississippi that interviewed 382 anglers. This survey indicated that approximately 60% of anglers who incidentally caught a sea turtle on hook-and-line reported it (i.e., 40% of anglers who incidentally caught a sea turtle did not report it) (Cook et al. 2016). It is important to note that in 2012 educational signs were installed at all fishing piers in Mississippi, alerting anglers to report accidental hook-and-line captures of sea turtles. After the signs were installed, there was a dramatic increase in the number of reported sea turtle hook-and-line captures. Though this increase in reported captures may not solely be related to outreach efforts, it does highlight the importance of educational signs on fishing piers. The STSSN in Mississippi indicated that inconsistency in reporting of captures may also be due to anglers' concerns over their personal liability, public perception at the time of the capture, or other consequences from turtle captures (M. Cook, STSSN, pers. comm. to N. Bonine, NMFS SERO PRD, April 17, 2015). Anglers often do not admit the incidental capture for fear of liability.

We believe it is most appropriate to use the unreported rate in the Cook et al. (2016) fishing pier study to estimate the future unreported captures at the Grand Isle State Park Fishing Pier. Because the study is in a similar location it is a reasonable proxy for reporting behavior at the Grand Isle State Park Fishing Pier. In addition, in the absence of additional information on factors that might affect angler reporting behavior, such as similarity of outreach and education, signage, or culture, we will assume fewer interactions were reported, as this will result in a higher total expected interactions. Therefore, we will address unreported captures by assuming that the expected annual reported captures of 3.1174 sea turtles per year at the Grand Isle State Park Fishing Pier represents 60% of the actual captures and 40% of sea turtle captures will be unreported. Reinitiation may be required if information reveals changes in reporting behavior.

$$\begin{aligned}&\mathit{Expected\ Annual\ Unreported\ Captures}\\&= (\mathit{Expected\ Annual\ Reported\ Captures} \div 60\%) \times 40\%\\&= (3.1174 \ \div 0.60) \times 0.40\\&= \ 2.0783\ \mathit{unreported\ captures\ per\ year}\ (\mathit{Table}\ 5, \mathit{Line}\ 2)\end{aligned}$$

6.3.3 Calculating Total Hook-and-Line Interactions with Sea Turtles

The number of captures in any given year can be influenced by sea temperatures, species abundances, fluctuating salinity levels in estuarine habitats where piers may be located, and other factors that cannot be predicted. For these reasons, we believe basing our future capture estimate on a 1-year estimated capture is largely impractical. Using our experience monitoring other fisheries, a 3-year time period is appropriate for meaningful evaluation of future impacts and monitoring. The triennial takes are set as 3-year running sums (i.e., 2025-2027, 2026-2028, 2027-2029, and so on) and not static 3-year periods (i.e., 2025-2027, 2028-2030, 2031-2033, and so on). This approach reduces the likelihood of reinitiation of the formal consultation process because of inherent variability in captures, while still allowing for an accurate assessment of how the proposed action is performing versus our expectations. Table 5 shows the projected total sea turtle captures at the consultation pier for any 3-year period based on the expected annual reported and unreported captures.

Table 5. Summary of Expected Reported and Unreported Captures of Sea Turtles at the Grand Isle State Park Fishing Pier

		Total
1.	Expected Annual Reported Captures	3.1174
2.	Expected Annual Unreported Captures	2.0783
Annual Total		5.1957
Triennial (3-year) Total		15.5870

6.3.4 Post Release Mortality of Sea Turtles

6.3.4.1 Estimating Post Release Mortality for Reported Captures

Almost all sea turtles that are captured, landed, and reported to the STSSN are evaluated by a trained veterinarian to determine if they can be immediately released alive or require a rehabilitation facility; exceptions may happen if the sea turtle breaks free before help can arrive. Sea turtles that are captured and reported to the STSSN may die onsite, may be evaluated, released alive, and subsequently suffer PRM later, or may be evaluated and taken to a rehabilitation facility. Those taken to a rehabilitation facility may be released alive at a later date or be kept in rehabilitation indefinitely (either due to serious injury or death). We consider those that are never returned to the wild population to have suffered PRM because they will never again contribute to the population. The risk of PRM to sea turtles from reported hook-and-line captures will depend on numerous factors, including how deeply the hook is embedded, whether the hook was swallowed, whether the sea turtle was released with trailing line, how soon and how effectively the hooked sea turtle was de-hooked or otherwise cut loose and released, and other factors which are discussed in more detail below.

We believe the 10-year STSSN dataset for offshore recreational hook and line captures and entanglements in Zone 12 is the most accurate representation of PRM for reported captures of sea turtles in the action area because this dataset pertains specifically to Mississippi where future reported captures are anticipated to occur. Table 6 provides a breakdown of final disposition of the 729 sea turtles caught or entangled in recreational hook-and-line gear in the STSSN dataset for Zone 12.

Table 6. Final Disposition of Sea Turtles from Reported Recreational Hook-and-Line Captures and Gear Entanglements in Zone 12, 2007-2016 (n=729)

	Dead or Died OnSite	Released Alive Immediately (Not Evaluated)	Released Alive, Immediately (Evaluated)	Taken to Rehab, Released Alive Later	Taken to Rehab, Kept of Died in Rehab
Number of Records	8	76	3	376	266
Percentage	1.1	10.4	0.4	51.6	36.5

Of the 729 sea turtles reported captured on recreational hook-and-line or entangled in gear in Zone 12, 37.6% were removed from the wild population either through death or being unable to be released from the rehabilitation facility (i.e., lethal captures, 1.1% + 36.5%) and 62.4% were released alive back into the wild population (i.e., non-lethal captures, 10.4 + 0.4 + 51.6).

To calculate the annual estimated lethal captures of reported sea turtles at the Grand Isle State Park Fishing Pier, we use the following equation:

$$
\begin{aligned}
&Annual\ Lethal\ Reported\ Captures \\
&= Expected\ Annual\ Reported\ Captures\ [Table\ 5, Line\ 1] \\
&\qquad\qquad \times Lethal\ Captures\ [calculated\ from\ Table\ 6] \\
&= 3.1174 \times 37.6\% \\
&= 1.1721\ per\ year\ \text{(Table 10, Line 1A)}
\end{aligned}
$$

To calculate the estimated annual non-lethal captures of reported sea turtles at the Grand Isle State Park Fishing Pier, we use the following equation:

$$\begin{aligned} &Annual\ Non-lethal\ Reported\ Captures \\ &= Expected\ Annual\ Reported\ Captures\ [Table\ 5, Line\ 1] \times Non- \\ &lethal\ Capture\ [calculated\ from\ Table\ 6] \\ &= 3.1174 \times 62.4\% \\ &= 1.9452\ per\ year \text{ (Table 10, Line 1B)} \end{aligned}$$

6.3.4.2 Estimating Post-Release Mortality for Unreported Hook-and-Line Interactions with Sea Turtles

Sea turtles that are captured and not reported to the STSSN may be released alive and subsequently suffer PRM. The risk of PRM to sea turtles from hook-and-line captures will depend on numerous factors, including how deeply the hook is embedded, whether the hook was swallowed, whether the sea turtle was released with trailing line, how soon and how effectively the hooked sea turtle was de-hooked or otherwise cut loose and released, and other factors which are discussed in more detail below. While the preferred method to release a hooked sea turtle safely is to bring it ashore and de-hook/disentangle it there and release it immediately, that cannot always be accomplished. The next preferred technique is to cut the line as close as possible to the sea turtle's mouth or hooking site rather than attempt to pull the sea turtle up to the pier. Some incidentally captured sea turtles are likely to break free on their own and escape with embedded/ingested hooks or trailing line. Because of considerations such as the tide, weather, and the weight and size of a hooked captured sea turtle, some will not be able to be de-hooked, and will be cut free by anglers and intentionally released. These sea turtles will escape with embedded or swallowed hooks, or trailing varying amounts of fishing line, which may cause post-release injury or death.

In January 2004, NMFS convened a workshop of experts to develop criteria for estimating PRM of sea turtles caught in the pelagic longline fishery based on the severity of injury. In 2006, those criteria were revised and finalized (Ryder et al. 2006). In February 2012, the Southeast Fisheries Science Center updated the criteria again by adding 3 additional hooking scenarios, bringing the total to 6 categories of injury (NMFS 2012). Table 7 describes injury categories for hardshell sea turtles captured on hook-and-line gear and the associated PRM estimates for sea turtles released with hook and trailing line greater than or equal to half the length of the carapace (i.e., Release Condition B as defined in (NMFS 2012). We use these criteria when estimating the PRM for unreported captures of sea turtles because it accounts for the expected differences in handling and care of reported versus unreported sea turtles. Please note the following, there is no PRM estimate of Release Condition B for Injury Category V. For Injury Category V, we believe it is prudent to use the PRM for Release Condition A (Released Entangled) because we know the sea turtle was released entangled without a hook, but we do not know how much line was remaining. For Injury Category 6, we believe it is prudent to use the PRM Release Condition D (Released with All Gear Removed) because we believe that if a fisher took the time to resuscitate the sea turtle, then it is likely the fisher also took the time to disentangle the animal completely before releasing it back into the wild.

Table 7. Estimated Post Release Mortality Based on Injury Category for Hardshell Sea Turtles Captured via Commercial Pelagic Longline and Released in Release Condition B (NMFS 2012).

Injury Category	Description	Post-release Mortality
I	Hooked externally with or without entanglement	20%
II	Hooked in upper or lower jaw with or without entanglement—includes ramphotheca (i.e., beak), but not any other jaw/mouth tissue parts	30%
III	Hooked in cervical esophagus, glottis, jaw joint, soft palate, tongue, or other jaw/mouth tissue parts not categorized elsewhere, with or without entanglement—includes all events where the insertion point of the hook is visible when viewed through the mouth.	45%
IV	Hooked in esophagus at or below level of the heart with or without entanglement—includes all events where the insertion point of the hook is not visible when viewed through the mouth	60%
V	Entangled only, no hook involved	50%
VI	Comatose/Resuscitated	60%

PRM varies based on the initial injury the animal sustained and the amount of gear left on the animal at the time of release. Again, we will rely on the STSSN dataset we used in Table 6 because this data includes on what part of the body the sea turtle was hooked for 629 of the 729 interactions (Table 8). SERO PRD assigned an Injury Category of 0 to all records with unknown hooking and entanglement locations. We exclude Injury Category 0 from the calculation because we are unsure of the location and therefore cannot assign a corresponding PRM. In this case, there are 100 interactions with an unknown hooking/entanglement location in the dataset.

Table 8. Category of Injury of Sea Turtles from Reported Recreational Hook-and-Line Captures and Gear Entanglements in Zone 12, 2007-2016 (n=629)

Injury Category	I	II	III	IV	V	VI
Number	50	10	458	94	17	0
Percentage	7.9	1.6	72.8	14.9	2.7	0

As above, we assume that 62.4% of the sea turtles captured at the Grand Isle State Park Fishing Pier will be reported, and that reported turtles will be sent to rehabilitation if needed. To estimate the fate of the 37.6% of sea turtles expected to go unreported at the Grand Isle State Park Fishing Pier, and therefore un-evaluated or rehabilitated, we use the estimated PRM for the injury categories in Table 7 along with the percentage of captures in each injury category in Table 8 to calculate the weighted PRM for each injury category. We then sum the weighted PRMs across all injury categories to determine the overall PRM for sea turtles (Table 9). This overall rate helps us account for the varying severity of future injuries and varying PRM associated with these injuries. Based on the assumptions we have made about the percentage of sea turtles that will be released alive without rehabilitation, the hooking location, and the amount of fishing gear likely to remain on an animal released immediately at the Grand Isle State Park Fishing Pier, we estimate a total weighted PRM of 45.1% for the 37.6% of sea turtles captured, unreported, and released immediately at the Grand Isle State Park Fishing Pier.

Table 9. Estimated Weighted and Overall Post Release Mortality for Sea Turtles Captured, Unreported, and Released Immediately

Injury Category	PRM (%) [from Table 6]	Percentage [from Table 8]	% Weighted PRM (% PRM × % Captures for each Injury Category)
I	20	7.9	1.6
II	30	1.6	0.5
III	45	72.8	32.8
IV	60	14.9	8.9
V	50	2.7	1.4
VI	60	0	0
Total Weighted PRM			45.1

To calculate the estimated annual lethal captures of unreported sea turtles at the Grand Isle State Park Fishing Pier, we use the following equation:

$Annual\ Unreported\ Lethal\ Captures$
$= Annual\ Unreported\ Captures\ [Table\ 4\ Line\ 2] \times Total\ Weighted\ PRM\ [Table\ 9]$
$= 2.0783 \times 45.1\%$
$= 0.9375\ per\ year$ (Table 10, Line 2A)

If the equation for calculating annual lethal captures of unreported sea turtles multiplies the annual unreported captures by the total weighted PRM of 45.1%, then the equation for calculating annual non-lethal captures of unreported sea turtles would multiply the annual unreported captures by 54.9% (100% – 45.1%). Therefore, to calculate the estimated annual non-lethal captures of unreported sea turtles at the Grand Isle State Park Fishing Pier, we use the following equation:

$Annual\ Unreported\ Non-lethal\ Unreported\ Captures$
$= Annual\ Unreported\ Captures\ [Table\ 5, Line\ 2] \times 54.9\%$
$= 2.0783 \times 54.9\%$
$= 1.1410\ per\ year$ (Table 10, Line 2B)

6.3.5 Calculating Total Post Release Mortality of Sea Turtles

As we discussed above, we use a 3-year running total to evaluate future impacts to sea turtles due to PRM. Table 10 shows the total sea turtle captures at Grand Isle State Park Fishing Pier for any 3-year consecutive period based on the expected annual lethal and non-lethal reported and unreported captures.

Table 10. Summary of Estimated Post Release Mortality of Sea Turtles

		A. Lethal	B. Non-lethal
1.	Annual Reported Captures	1.1721	1.9452
2.	Annual Unreported Captures	0.9375	1.1410
Annual Total		2.1096	3.0862
Triennial (3-year) Total		6.3288	9.2586

6.4 Estimating Hook-and-Line Captures by Species of Sea Turtle

Of the sea turtles in the STSSN Zone 12 data identifiable to species and which may be adversely affected by the proposed action (n=662), 0.8% were green (n=5), 98.2% were Kemp's ridley (n=650), and 1.1% were loggerhead sea turtles (n=7) (Table 2). We will assume the same potential species composition for future captures at the Grand Isle State Park Fishing Pier because these are the best available data regarding the relative abundance of sea turtle species that may be affected by hook and line gear in the action area. In the absence of any reported capture data for the Grand Isle State Park Fishing Pier, we believe that using the complete dataset for Zone 12 is a more accurate representation of which sea turtle species could be in the action area and affected by recreational hook and line interactions at the Grand Isle State Park Fishing Pier given the variability in the species presence in the general area. Thus, the entire Zone 12 dataset accounts for the captures previously reported, while also providing a basis to estimate take of other sea turtles species that are expected to be present in the area and affected by hook and line gear. Table 11 estimates the number of lethal and non-lethal captures by sea turtles species for any consecutive 3-year period based on our calculations from Sections 6.3.1 and 6.3.2. Because the calculated estimates of sea turtle captures are fractions, we round the numbers of captures up to the nearest whole number. While this results in an increase in the total number of sea turtles, compared to what is presented in the non-species-specific total estimates in Table 5 and Table 10, this approach ensures that we are adequately analyzing the effects of the proposed action on whole animals, and that impacts from the proposed action can be more easily tracked. The impacts of future captures to the individual green sea turtle DPSs are discussed in the Jeopardy Analysis (Section 8) and presented in the Incidental Take Statement (Section 10).

Table 11. Estimated Captures of Sea Turtle Species at the Grand Isle State Park Fishing Pier for Any Consecutive 3-Year Period

Species	Lethal	Non-lethal	Total
Green sea turtle (NA DPS)	1 (0.008 × 6.3288 = 0.0506)	1 (0.008 × 9.2586 = 0.0741)	Up to 2
Kemp's ridley sea turtle	7 (0.982 × 6.3288 = 6.2149)	10 (0.982 × 9.2586 = 9.0919)	Up to 17
Loggerhead sea turtle (NWA DPS)	1 (0.011 × 6.3288 = 0.0696)	1 (0.011 × 9.2586 = 0.1018)	Up to 2

6.5 Estimating Fishing Gear Interactions with Giant Manta Ray

Based upon our professional judgement, we believe the best available data to estimate future observed fishing interactions with giant manta ray at public fishing structures come from the surveys conducted by the Marine Megafauna Foundation (MMF). In 2016, the MMF began conducting aerial and boat-based surveys between St. Lucie Inlet and Boynton Beach Inlet on the east coast of Florida in Palm Beach County, a known area of high abundance for juvenile giant manta ray (Pate and Marshall 2020). During survey efforts researchers documented high occurrences of recreational fishing interactions with giant manta ray (i.e., foul hooked or entangled) (Pate and Marshall 2020; Pate et al. 2022). According to the information provided by the NOAA RC and the applicant, there have been no reported captures of or interactions with giant manta ray at the Grand Isle State Park Fishing Pier. In the absence of data specific to areas adjacent to or within the action area, we believe the MMF survey data is the best available for calculating the estimated number of future observed fishing gear interactions with giant manta ray at the Grand Isle State Park Fishing Pier.

Between 2016 and 2022, MMF documented 58 interactions between fishing gear and giant manta ray within the survey area (J. Pate, MMF, unpublished data). Entangled or foul-hooked giant manta rays typically were observed within an average of 1.2 mi (2.0 km) from a fishing pier or inlet (J. Pate, MMF, unpublished data). We assume that all giant manta rays observed entangled or foul-hooked during these surveys occurred from fishing piers due to their close proximity to fishing piers and the fact that individuals had multiple fishing gear interactions within the survey area.

In the MMF survey area (i.e., between St. Lucie Inlet and Boynton Beach Inlet, Palm Beach County, Florida), there are 4 public ocean-facing fishing structures – Jupiter Inlet, Juno Beach Pier, Lake Worth Pier, and Boynton Beach Inlet. These piers are similar in size and location (i.e., relatively large, public ocean facing or inlet fishing structures), and have similar angler effort. Pate et al. (2020) conducted semi-structured surveys to assess recreational anglers' knowledge of and attitudes toward giant manta ray. These surveys revealed anglers fishing from these locations use similar baits, equipment, and fishing techniques. Therefore, we believe that the potential for interactions with giant manta ray is likely the same at all 4 piers in the MMF survey area.

To calculate the average number of observed interactions with fishing gear within the MMF survey area, we use the available MMF data and the following equation:

Average Interactions Per Structure in 7 years
= Sum of Reported Interactions in 7 years ÷ 4 locations
= 58 ÷ 4
= 14.5 per structure in 7 years

To calculate the estimated expected annual number of observed fishing gear interactions with giant manta ray at the Grand Isle State Park Fishing Pier, we refer to the MMF data above and use the following equation:

Expected Annual Interactions
= Average Reported Interactions Per Structure in 7 years ÷ 7 years
= 14.5 ÷ 7
= 2.07 interactions per structure per year

While the survey occurred in a known area of high juvenile abundance, it is the best available data we have to estimate giant manta ray interactions that may occur at the consultation pier. Because the calculated estimate is a fraction, we round the number of interactions per structure per year up to the nearest whole number to get a total of 3 observed fishing gear interactions per structure per year. As discussed above, we believe using a 3-year period is appropriate for meaningful monitoring. Therefore, up to 9 interactions with giant manta ray at the consultation pier may occur in any consecutive 3-year period. As previously stated, we believe that all captures of giant manta ray will be non-lethal with no associated PRM.

7 CUMULATIVE EFFECTS

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

8 JEOPARDY ANALYSIS

8.1 Jeopardy Analysis

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

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

The analyses conducted in the previous sections of this Opinion serve to provide a basis to determine whether the proposed action would be likely to jeopardize the continued existence of green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and the giant manta ray. In Section 6.0, we outlined how the proposed action can adversely affect these species. Now we turn to an assessment of the species response to these impacts, in terms of overall population effects, and whether those effects of the proposed action, when considered in the context of the Status of the Species (Section 4.0), and the Environmental Baseline (Section 5.0), will jeopardize the continued existence of the affected species. For any species listed globally, our jeopardy determination must evaluate whether the proposed action will appreciably reduce the likelihood of survival and recovery at the species' global range. For any species listed as DPSs, a jeopardy determination must evaluate whether the proposed action will appreciably reduce the likelihood of survival and recovery of that DPS.

8.2 Green Sea Turtles (North Atlantic DPS)

8.2.1 Survival

The proposed action is expected to result in capture of up to 2 green sea turtles (1 lethal, 1 non-lethal) from the North Atlantic DPS over any consecutive 3-year period. Any potential non-lethal capture during any consecutive 3-year period are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individual suffering non-lethal injuries or stresses is expected to fully recover such that no reductions in reproduction or numbers of green sea turtles are anticipated. The non-lethal captures will occur in the action area, which encompass a small portion of the overall range or distribution of green sea turtles within the North Atlantic DPS. Any incidentally caught animals would be released within the general area where caught and no change in the distribution of North Atlantic DPS green sea turtles would be anticipated. The potential lethal captures during any consecutive 3-year period would reduce the number of North Atlantic DPS green sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. A lethal capture would also result in a reduction in future reproduction, assuming the individual was female and would have survived otherwise to reproduce. For example, as discussed in this Opinion, an adult green sea turtle can lay up to 7 clutches (usually 3-4) of eggs every 2-4 years, with a mean clutch size of 110-115 eggs per nest, of which a small percentage is expected to survive to sexual maturity. The potential lethal captures are expected to occur in a small, discrete area and green sea turtles in the North Atlantic DPS 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 the species likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In the Status of Species (Section 4.2.1), we presented the status of the North Atlantic DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the Environmental Baseline, we outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect the North Atlantic DPS. In the Cumulative Effects, we discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area.

In Section 4.2.2, we summarized the available information on number of green sea turtle nesters and nesting trends at North Atlantic DPS beaches. In the absence of any total population estimates, nesting trends are the best proxy for estimating population changes. Eight nesting sites have high levels of abundance (i.e., < 1000 nesters), located in Costa Rica, Cuba, Mexico, and Florida. Tortuguero, Costa Rica is by far the predominant nesting site, accounting for an estimated 79% of nesting for the DPS (Seminoff et al. 2015). A recent long-term study spanning over 50 years of nesting at Tortuguero found that while nest numbers increased steadily over 37 years from 1971-2008, the rate of increase slowed gradually from 2000-2008. After 2008 the nesting trend has been downwards, with current nesting levels having reverted to that of the mid 1990's and the overall long-term trend has now become negative (Restrepo, et al. 2023).

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). Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). While nesting in Florida has shown dramatic increases over the past decade, individuals from the Tortuguero, the Florida, and the other Caribbean and Gulf of America populations in the North Atlantic DPS intermix and share developmental habitat. Therefore, threats that have affected the Tortuguero population as described previously, may ultimately influence the other population trajectories, including Florida. Given the large size of the Tortuguero nesting population, which is currently in decline, its status and trend largely drives the status of North Atlantic DPS.

Aside from the long-term increasing nesting trend observed in Florida, the declining trend in nesting observed in Tortuguero indicates a species in decline. However, since the proposed project is anticipated to result in the potential lethal take of up to 1 green sea turtle from the North Atlantic DPS during any consecutive 3-year period, which is only a small fraction of the reduced but still large overall nesting population, and we have no reason to believe nesting females will be disproportionately affected, we believe the potential mortality associated with the proposed action will have no detectable effect on current nesting trends.

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

8.2.2 Recovery

The North Atlantic 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 1991b) does exist. Since the animals within the North Atlantic 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 North Atlantic DPS, is developed. The Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

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

According to data collected from Florida's index nesting beach survey from 1989-2021, green sea turtle nest counts across Florida index beaches have increased substantially from a low of approximately 267 in the early 1990s to a high of almost 41,000 in 2019 (See Figure 3). Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011. The pattern departed from the low lows and high peaks in 2020 and 2021 as well, when 2020 nesting only dropped by half from the 2019 high, while 2021 nesting increased over the 2020 nesting, with another increase in 2022 still well below the 2019 high. This indicates that the first listed recovery objective is 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 increased, which is consistent with the criteria of the second listed recovery objective.

The potential lethal take of up to 1 green sea turtle from the North Atlantic DPS during any consecutive 3-year period will result in a reduction in numbers when a capture occurs; however, it is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. The non-lethal take of 1 green sea turtles from the North Atlantic DPS would not affect the adult female nesting population or number of nests per nesting season. Thus, the proposed action will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of North Atlantic DPS green sea turtles' recovery in the wild.

8.2.3 Conclusion

The combined potential lethal and non-lethal take of green sea turtles from the North Atlantic DPS associated with the piers is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the North Atlantic DPS of green sea turtle in the wild.

8.3 Kemp's Ridley Sea Turtle

8.3.1 Survival

The proposed action is expected to result in the capture of up to 17 Kemp's ridley sea turtles (7 lethal, 10 non-lethal) during any consecutive 3-year period. Any potential non-lethal capture is not expected to have any measurable impact on the reproduction, numbers, or distribution of the species. The individual suffering non-lethal injuries or stresses are expected to fully recover such that no reductions in reproduction or numbers of Kemp's ridley sea turtles are anticipated. A non-lethal capture will occur in the action area, which encompasses a small portion of this species overall range/distribution. Any incidentally caught animal would be released within the general area where caught and no change in the distribution of Kemp's ridley sea turtles would be anticipated. The potential lethal captures during any consecutive 3-year period would reduce the species' population compared to the number that would have been present in the absence of the proposed actions, assuming all other variables remained the same. The Turtle Expert Working Group (TEWG 1998b) estimates age at maturity from 7-15 years for this species. Females return to their nesting beach about every 2 years (TEWG 1998b). The mean clutch size for Kemp's ridley sea turtle is 100 eggs/nest, with an average of 2.5 nests per female per season. Lethal take 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 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. However, the potential lethal take during any consecutive 3-year period is expected to occur in a small, discrete area and Kemp's ridley sea turtle generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In the Status of Species, we presented the status of the Kemp's ridley sea turtle, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the Environmental Baseline, we considered the past and present impacts of all state, federal, or private actions and other human activities in, or having effects in, the action areas that have affected and continue to affect this DPS.

In the absence of any total population estimates for Kemp's ridley sea turtle, nesting trends are the best proxy for estimating population changes. 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 trend. In Section 4.2.3, we summarized the available information on Kemp's ridley sea turtle nesters and nesting trends. At this time, it is unclear whether the increases and declines in Kemp's ridley sea turtle nesting seen over the past decade at nesting beaches in Mexico, or the similar trend with the emerging Texas population, represents a population oscillating around an equilibrium point or if nesting will decline or increase in the future. With the recent period of increases in nesting (2015-2017) bookended by recent periods of declining numbers of nests (2013-2014 and 2018-2019), it is too early to tell whether the long-term trend line is affected. So at this point we can only conclude that the population has dramatically rebounded from the low lows in the 80's and 90's, but we cannot ascertain a current population trend or trajectory. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals and that the potential combined lethal take of up to 7 Kemp's ridley sea turtles during any consecutive 3-year period attributed to the proposed action will not have any measurable effect on that trend. After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe that recreational fishing from the Grand

Isle State Park Fishing Pier is not reasonably expected to cause an appreciable reduction in the likelihood of survival of Kemp's ridley sea turtle in the wild.

8.3.2 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. 2011b) lists the following relevant recovery 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.*

The recovery plan states the average number of nests per female is 2.5; it sets a recovery goal of 10,000 nesting females associated with 25,000 nests. The 2012 nesting season recorded approximately 22,000 nests. Yet, in 2013 through 2014, there was a second significant decline, with only 16,385 and 11,279 nests recorded, respectively, which would equate to 6,554 nesting females in 2013 (16,385 / 2.5) and 4,512 in 2014 (11,279 / 2.5). Nest counts increased in the last three years, but they did not quite reach 25,000 in 2017 and slightly declined in 2018; however, it is clear that the population trend has been increasing over the last 2 decades. The increase in Kemp's ridley sea turtle nesting 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 U.S., and possibly other changes in vital rates (TEWG 1998a; TEWG 2000).

The potential lethal take of up to 7 Kemp's ridley sea turtle during any consecutive 3-year period by recreational fishing at the pier will result in a reduction in numbers and reproduction; however, it is unlikely to have any detectable influence on the nesting trends noted above. Given that annual nest numbers are in the thousands, the projected loss is not expected to have any discernable impact to the species. Thus, recreational fishing at the Grand Isle State Park Fishing Pier will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of the Kemp's ridley sea turtles' recovery in the wild.

8.3.3 Conclusion

The combined lethal and non-lethal take of Kemp's ridley sea turtles associated with the Grand Isle State Park Fishing Pier is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of Kemp's ridley sea turtle in the wild.

8.4 Loggerhead Sea Turtle (Northwest Atlantic DPS)

8.4.1 Survival

The proposed action is expected result in the capture of up to 2 loggerhead sea turtles (1 lethal, 1 nonlethal) from the Northwest DPS during any consecutive 3-year period. Any potential non-lethal captures during any consecutive 3-year period are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individual suffering non-lethal injuries or stresses is expected to fully recover such that no reductions in reproduction or numbers of green sea turtles are anticipated. All non-lethal captures will occur in the action area, which encompass a small portion of the overall range or distribution of loggerhead sea turtles within the Northwest Atlantic DPS. Any incidentally caught animals would be released within the general area where caught and no change in the distribution of Northwest Atlantic DPS of loggerhead sea turtles would be anticipated. The potential lethal captures during any consecutive 3-year period would reduce the number of Northwest Atlantic loggerhead sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. Potential lethal captures would 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. Thus, the loss of adult females could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. However, a reduction in the distribution of loggerhead sea turtles is not expected from lethal take attributed to the piers. The anticipated lethal take is expected to occur in a small, discrete area, and loggerhead sea turtles in the Northwest Atlantic DPS generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

In the absence of any total population estimates, nesting trends are the best proxy for estimating population changes. Abundance estimates in the western North Atlantic indicate the population is large (i.e., several hundred thousand individuals). In Section 4.2.4, we summarized available information on number of loggerhead sea turtle nesters and nesting trends. Nesting trends across all of the recovery units have been steady or increasing over several years against the background of the past and ongoing human and natural factors that have contributed to the current status of the species. Additionally, in-water research suggests the abundance of neritic juvenile loggerheads is steady or increasing.

The proposed action could lethally take up to 1 loggerhead sea turtle during any consecutive 3-year period. While the potential lethal capture of up to 1 loggerhead sea turtle during any consecutive 3-year period will affect the population, in the context of the overall population's size and current trend, we do not expect this loss to result in a detectable change to the population numbers or increasing trend. After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the Northwest Atlantic DPS of loggerhead sea turtle in the wild.

8.4.2 Recovery

The recovery plan for the for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2009) was written prior to the loggerhead sea turtle DPS listings. However, this plan deals with the populations that comprise the current NWA DPS and is therefore, the best information on recovery criteria and goals for the DPS. It lists the following recovery objectives that are relevant to the effects of the proposed actions:

- *Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females*
- *Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes*

Recovery is the process of removing threats so self-sustaining populations persist in the wild. The proposed actions would not impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy. The recovery plan estimates that the population will reach recovery in 50-150 years following implementation of recovery actions. 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.

In Section 4.2.4, we summarized available information on number of loggerhead sea turtle nesters and nesting trends. Nesting trends across all of the recovery units have been steady or increasing over several years against the background of the past and ongoing human and natural factors that have contributed to the current status of the species. FWRI examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but non-significant increasing trend. Looking at the data from 1989 through 2016, FWRI concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability from 2012-2016 resulting in widening confidence intervals. Nesting at the core index beaches declined in 2017 to 48,033, and rose again each year through 2020, reaching 53,443 nests, dipping back to 49,100 in 2021, and then in 2022 reaching the second-highest number since the survey began, with 62,396 nests. It is important to note that with the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.), it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani et al. 2019). In-water research suggests the abundance of neritic juvenile loggerheads is also steady or increasing. Thus, the potential lethal capture of up to 1 loggerhead sea turtle during any consecutive 3-year periods is so small in relation to the overall population, that it would be undetectable, 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. The potential non-lethal capture from the Northwest Atlantic DPS would not affect the adult female nesting population, number of nests per nesting season, or juvenile in-water populations. Thus, recreational fishing at the Grand Isle State Park Fishing Pier will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of the Northwest Atlantic DPS of loggerhead sea turtles' recovery in the wild.

8.4.3 Conclusion

The combined potential lethal and non-lethal captures during any consecutive 3-year period of loggerhead sea turtles associated with the Grand Isle State Park Fishing Pier is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the Northwest Atlantic DPS of the loggerhead sea turtle in the wild.

8.5 Giant Manta Ray

The proposed action is expected to result in the capture of 9 giant manta rays over any consecutive 3-year period. We expect all captures to be non-lethal with no associated PRM.

8.5.1 Survival

The non-lethal capture of giant manta ray over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals captured are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur in the small, discrete action area and would be released within the general area where caught, no change in the distribution of giant manta ray is anticipated.

8.5.2 Recovery

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

> 1) Stabilize population trends through reduction of threats, such that the species is no longer declining throughout a significant portion of its range; and
>
> 2) Gather additional information through research and monitoring on the species' current distribution and abundance, movement and habitat use of adult and juveniles, mortality rates in commercial fisheries (including at-vessel and PRM), and other potential threats that may contribute to the species' decline.

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

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

8.5.3 Conclusion

The potential non-lethal capture over any consecutive 3-year period associated with the proposed action is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of giant manta ray in the wild. Mortalities are not expected, and the proposed action furthers outreach efforts by ensuring signs are maintained at the pier to educate anglers about safe handling and reporting interactions with the species. Thus, the recreational fishing effects from the proposed pier will not result in an appreciable reduction in the likelihood of giant manta ray recovery in the wild.

9 CONCLUSION

We reviewed the Status of the Species, the Environmental Baseline, the Effects of the Action, and the Cumulative Effects using the best available data.

The proposed action will result in the take of up to 2 green sea turtles (North Atlantic DPS), 17 Kemp's ridley sea turtles, 2 loggerhead sea turtles (Northwest Atlantic DPS) (Table 14), and 9 giant manta ray over the course of any given 3 consecutive year period. Given the nature of the proposed action and the information provided above, we conclude that the action, as proposed, is not likely to jeopardize the continued existence of green sea turtle (North Atlantic DPS), Kemp's ridley sea turtles, loggerhead sea turtles (Northwest Atlantic DPS), and giant manta ray.

10 INCIDENTAL TAKE STATEMENT

10.1 Overview

Section 9 of the ESA and protective regulations issued pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. *Take* is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct (ESA Section 2(19)). *Incidental take* refers to takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant. Under the terms of section 7(b)(4) and section 7(o)(2), taking that would otherwise be considered prohibited under section 9 or section 4(d) but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA, provided that such taking is in compliance with the Reasonable and Prudent Measures and the Terms and Conditions of the Incidental Take Statement of the Opinion.

The take of giant manta ray by the proposed action is not prohibited under ESA Section 9, as no Section 4(d) Rules for the species have been promulgated. However, a circuit court case held that non-prohibited incidental take must be included in the Incidental Take Statement (*CBD v. Salazar*, 695 F.3d 893 [9th Circuit 2012]). Though the *Salazar* case is not a binding precedent for this action, which occurs outside of the 9th Circuit, NMFS finds the reasoning persuasive and is following the case out of an abundance of caution and because we anticipate that the ruling will be more broadly followed in future cases. Providing an exemption from Section 9 liability is not the only important purpose of specifying take in an Incidental Take Statement. Specifying incidental take ensures we have a metric against which we can measure whether reinitiation of consultation is required. Including these species in the Incidental Take Statement also ensures that we identify Reasonable and Prudent Measures that we believe are necessary or appropriate to minimize the impact of the incidental take on the species.

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

As soon as NOAA RC (the applicant) becomes aware of any take of an ESA-listed species under NMFS' purview that occurs during the proposed action, the applicant shall report the take to NMFS SERO PRD via the NMFS SERO Endangered Species Take Report Form (https://forms.gle/85fP2da4Ds9jEL829). This form shall be completed for each individual known reported capture, entanglement, stranding, or other take incident. Information provided via this form shall include the title, Grand Isle State Park Improvement, the issuance date, and ECO tracking number (SERO-2024-02183) for this Opinion; the species name; the date and time of the incident; the general location and activity resulting in capture; condition of the species (i.e., alive, dead, sent to rehabilitation); size of the individual, behavior, identifying features (i.e., presence of tags, scars, or distinguishing marks), and any photos that may have been taken. At that time, consultation may need to be reinitiated.

The NOAA RC has a continuing duty to ensure compliance with the reasonable and prudent measures and terms and conditions included in this Incidental Take Statement. If the NOAA RC (1) fails to assume and implement the terms and conditions or (2) fails to require the terms and conditions of the Incidental Take Statement through enforceable terms that are added to the permit or grant document or other similar document, the protective coverage of Section 7(o)(2) may lapse. In order to monitor the impact of incidental take, NOAA RC must report the progress of the action and its impact on the species to NMFS as specified in the Incidental Take Statement (50 CFR 402.14(i)(4)).

10.2 Amount of Extent of Anticipated Incidental Take

Based on the above information and analyses, NMFS believes that the proposed action is likely to adversely affect the green sea turtle (North Atlantic DPS), Kemp's ridley sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), and giant manta ray. These effects will result from incidental capture from recreational fishing from the proposed Grand Isle State Park Fishing Pier as described in Section 6 (i.e. entanglement and hooking by recreational fishing gear, and trailing line). NMFS anticipates the following incidental take may occur as a result of the proposed action over any consecutive 3-year period (i.e., 2025-2027, 2026-2028, 2027-2029, and so on) (**Table 12**).

The take limits prescribed in this Opinion that will trigger the requirement to reinitiate consultation must be based on the amount of take that we expect to be *reported* as it will be impossible to count the incidents that go unreported. The best available information for estimating the future take if sea turtles and giant manta ray that will be reported at the Grand Isle State Park Fishing Pier is described in Section 6.

In Section 6.3.3 we developed an estimate of the total number of sea turtle captures expected to be reported annually (3.1174; Table 5, line 1). We take that number and multiply by 3 to get the 3-year total estimate of reported sea turtle captures (3.1174 x 3 = 9.3522). We then apply that number to the species breakdown reported in the Mississippi STSSN data for recreational hook-and-line captures and gear entanglements in Zone 12 (0.8% green; 98.20% Kemp's ridley; and 1.1% loggerhead) to the 3-year total to estimate of reported take of each species (Table 12). For those estimates that come out to be less than 1, we round up to 1 to reach a whole number that can be used as a take limit. The anticipated, unreported sea turtle takes are not directly monitored but can be estimated from reported takes using the process described in Section 6.3.2. Based on the data collected from the Hill (2013) fishing pier study, we anticipate 92% of sea turtle take will go unreported.

Table 12. Incidental Take Limits by Species for Any Consecutive 3-Year Period.

Species	**Total Estimated Reported Captures**	**Incidental Take Limits that will Trigger Reinitiation**
Green sea turtle (North Atlantic DPS)	9.3522 x 0.008 = 0.0748, rounded up to 1	No more than 1 reported capture
Kemp's ridley sea turtle	9.3522 x 0.982 = 9.1838, rounded up to 10	No more than 10 reported captures
Loggerhead sea turtle (Northwest Atlantic DPS)	9.3522 x 0.011 = 0.1029, rounded up to 1	No more than 1 reported capture
Giant manta ray	NA	No more than 9 reported captures

It is important to note that the mortality rates estimated in Section 6.3.4 are not likely to be detected in the initial reporting of captures, as most sea turtles are expected to live for some period of time following capture. Some of these individuals may be sent to rehabilitation facilities and later die in those facilities, or may be released and die in the wild from undetected injuries, as discussed in our PRM analysis in Section 6.3.4. While it is also possible that some sea turtles may die immediately from severe injuries related to hook-and-line capture or entanglement (which will be included in the annual reports discussed below), we do not expect that result (see Section 6.3.4.1). At the time of the interaction, we expect sea turtle take in the above ITS to be non-lethal. As discussed in Section 6.3.4.1, above, up to 31.65% of the reported interactions could result in a mortality, and reports of such PRM are consistent with the analysis in this Opinion and this ITS. Likewise, we expect PRM of the unreported sea turtle interactions, as described in Section 6.3.4.1.

Again, we expect all interactions with giant manta ray to be non-lethal with no associated PRM.

10.3 Effect of Take

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

10.4 Reasonable and Prudent Measures

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

The Reasonable and Prudent Measures and terms and conditions are required to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species (50 CFR 402.14(i)(1)(ii) and (iv)). These measures and terms and conditions must be implemented by the NOAA RC for the protection of section 7(o)(2) to apply. The NOAA RC has a continuing duty to ensure compliance with the reasonable and prudent measures and terms and conditions included in this Incidental Take Statement. If the NOAA RC fails to adhere to the terms and conditions of the Incidental Take Statement through enforceable terms, or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse. To monitor the impact of the incidental take, the NOAA RC must report the progress of the action and its impact on the species to SERO PRD as specified in the Incidental Take Statement [50 CFR 402.14(i)(4)].

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

1. The NOAA RC must ensure that the applicant provides take reports regarding all reported interactions with ESA-listed species at Grand Isle State Park Fishing Pier.
2. The NOAA RC must ensure that the applicant minimizes the likelihood of injury or mortality to ESA-listed species resulting from hook-and-line capture or entanglement by activities at Grand Isle State Park Fishing Pier.
3. The NOAA RC must ensure that the applicant reduces the impacts to incidentally captured ESA-listed species.
4. The NOAA RC must ensure that the applicant coordinates periodic fishing line and underwater debris removal (i.e., cleanup) events with non-governmental or other local organizations.
5. The NOAA RC must ensure that the applicant conducts quarterly angler surveys that solicit information on sea turtle encounters.

10.5 Terms and Conditions

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

The following T&Cs implement the above RPMs:

1. To implement RPM No. 1, No. 4, and No. 5, NOAA RC shall include a special condition of the grant that directs the grant applicant to reports all known hook-and-line captures of ESA-listed species and any other takes of ESA-listed species at Grand Island Fishing Pier to the NMFS SERO PRD and NOAA RC.
    a. If and when the applicant becomes aware of any known reported capture, entanglement, stranding, or other take, the applicant must report it to NMFS SERO PRD via the NMFS SERO Endangered Species Take Report Form (https://forms.gle/85fP2da4Ds9jEL829).
        i. The applicant shall complete the SERO PRD reporting form using the NMFS SERO Endangered Species Take Report Form (https://forms.gle/85fP2da4Ds9jEL829).
        ii. This form shall be completed for each individual known reported capture, entanglement, stranding, or other take incident.
        iii. The form must include the species name, state the species, date and time of the incident, general location and activity resulting in capture (i.e., alive, dead, sent to rehabilitation), size of the individual, behavior, identifying features (i.e., presence of tags, scars, or distinguishing marks), and any photos that may have been taken.
        iv. Emails must reference the project name (Grand Isle State Park Improvement) and tracking number (SERO-2024-02183) in the subject line.
        v. The email must state the species, date and time of the incident, general location and activity resulting in capture (i.e., fishing from Grand Island Fishing Pier by hook-and-line), condition of the species (i.e., alive, dead, sent to rehabilitation), size of the individual, behavior, identifying features

(i.e., presence of tags, scars, or distinguishing marks), and any photos that may have been taken.

b. Every year, the applicant must submit a summary report of capture, entanglement, stranding, or other take of ESA-listed species to NMFS SERO PRD by email: nmfs.ser.esa.consultations@noaa.gov.
    i. All emails and summary reports must reference this Opinion by the NMFS tracking number for this Opinion (SERO-2024-02183 Grand Isle State Park Improvement Project) and date of issuance.
    ii. The summary report will contain the following information: the total number of ESA-listed species captures, entanglements, strandings, or other take that was reported at or adjacent to the pier included in this Opinion.
    iii. The report will contain all information for any sea turtles taken to a rehabilitation facility holding an appropriate U.S. Fish and Wildlife Native Endangered and Threatened Species Recovery permit. This information can be obtained from the appropriate State Coordinator for the STSSN (https://www.fisheries.noaa.gov/state-coordinators-sea-turtle-stranding-and-salvage-network)
    iv. The summary report shall be submitted even when there has been no reported take of ESA-listed species.
    v. The first report will be submitted by January 31, 2026 and cover the period from the pier opening until December 31, 2025. The second report will be submitted the following January 31, and cover the previous calendar year and the information in the first report. Thereafter, reports will be prepared every year, covering the prior rolling three-year period, and emailed no later than January 31 of any year.
    vi. Summary reports will include current photographs of signs and bins required in T&Cs No. 2 below; information on cleaning event conducted in that year, required in T&C No. 3, below; and the results from the quarterly angler surveys conducted in that year, required in T&C No. 4, below.

2. To implement RPMs No. 2 and 3, NOAA RC shall include a special condition of the grant that directs the applicant or its designated entity carrying out the project to:
    a. Install and maintain the following NMFS Protected Species Educational Signs: “Save Dolphins, Sea Turtles, Sawfish, and Manta Ray; Do Not Catch or Harass Sea Turtles, Report a Sturgeon.” The signs must also include instructions for contacting park officials trained in landing, handling and de-hooking sea turtles, described in T&C 2.c., below.
        i. Signs will be posted at the entrance to and at the terminal end of the pier, and at regular intervals along the pier. In all instances, the view of these signs is unobstructed.
        ii. Signs will be installed prior to opening the pier for public use.
        iii. Photographs of the installed signs will be emailed to NMFS’s Southeast Regional Office by email (nmfs.ser.esa.consultations@noaa.gov) with the

NMFS tracking number for this Opinion (SERO-2024-02183 Grand Isle State Park Improvement Project) and date of issuance.

iv. Sign designs and installation methods are provided at the following website: https://www.fisheries.noaa.gov/southeast/consultations/protected-species-educational-signs.

v. Additionally, current photographs of the signs will be included in each report required by T&C No. 1.b., above.

b. Install and maintain monofilament recycling bins and trash receptacles at the piers to reduce the probability of trash and debris entering the water.

   i. Monofilament recycling bins and trash receptacles will be installed prior to opening the pier for public use.

   ii. Photographs of the installed bins will be emailed to NMFS's Southeast Regional Office by email (nmfs.ser.esa.consultations@noaa.gov) with the NMFS tracking number for this Opinion (SERO-2024-02183 Grand Isle State Park Improvement Project) and date of issuance.

   iii. The applicant must regularly empty the bins and trash receptacles and make sure they are functional and upright.

   iv. Additionally, current photographs of the bins will be included in each report required by T&C No. 1.b., above.

c. Ensure that one Grand Isle State Park employee is "on call" during the park's normal operating hours and available to assist in the landing and handling of any turtles captured at the pier.

   i. Instructions for contacting trained Grand Isle State Park officials will be included on the signage at the piers (T&C 2.a.).

   ii. Each official will complete training on landing, handling and de-hooking sea turtles.

   iii. Trained Grand Isle State Park officials will have ready access to sea turtle landing and de-hooking equipment (long handled nets, pliers, etc.) as well as current contact information for local turtle rescue and rehabilitation facilities.

3. To implement RPM No. 2 and No. 4, NOAA RC shall include a special condition of the grant that directs the applicant or its designated entity carrying out the project to:
   a. Conduct annual pier cleanup to remove derelict fishing line and associated gear and other debris and trash from around the pier structure.
   b. Submit a record of each cleaning event in the report required by T&C No. 1.b. above.
4. To implement RPM No. 5, NOAA RC must ensure that angler surveys are conducted at the Grand Isle State Park Fishing Pier on a quarterly basis.
   a. These surveys shall include questions on sea turtle and giant manta ray encounters at the Grand Isle State Park Fishing Pier.
   b. Supplemental educational material on sea turtle rescue and reporting shall be distributed with the surveys.

11 CONSERVATION RECOMMENDATIONS

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

Sea turtles:

- Conduct or fund research that investigates ways to reduce and minimize mortality of ESA-listed sea turtles in the recreational hook-and-line fishery.
- Conduct or fund outreach designed to increase the public's knowledge and awareness of ESA-listed sea turtle species.

Giant manta rays:

- Conduct or fund outreach designed to increase the public's knowledge and awareness of giant manta ray.

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.

12 REINITIATION OF CONSULTATION

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

13 LITERATURE CITED

81 FR 20057. 2016. Endangered and Threatened Wildlife and Plants; Final Rule To List Eleven Distinct Population Segments of the Green Sea Turtle (*Chelonia mydas*) as Endangered or Threatened and Revision of Current Listings Under the Endangered Species Act. Final Rule. Federal Register 81(66):20057 -20090.

Ackerman, R. A. 1997. The nest environment and the embryonic development of sea turtles. Pages 83-106 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.

Adams, D. H., and E. Amesbury. 1998. Occurrence of the manta ray, Manta birostris, in the Indian River Lagoon, Florida. Florida Scientist 61(1):7-9.

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

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

Aguilar, R., J. Mas, and X. Pastor. 1994. Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle *Caretta caretta* population in the western Mediterranean. Pages 91-96 *in* J. I. Richardson, and T. H. Richardson, editors. Proceedings of the 12th Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Jekyll Island, Georgia.

Aguilar, R., J. Mas, and X. Pastor. 1995. Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle, Caretta caretta, population in the western Mediterranean. Pages 1 *in* 12th Annual Workshop on Sea Turtle Biology and Conservation, Jekyll Island, Georgia.

Aguirre, A., G. Balazs, T. 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.

Alava, M.N.R., Dolumbaló, E.R.Z., Yaptinchay, A.A. and Trono, R.B. 2002. Fishery and trade of whale sharks and manta rays in the Bohol Sea, Philippines. Pp. 132-148. In: S.L. Fowler, T.M. Reed and F.A. Dipper (eds), Elasmobranch Biodiversity, Conservation and Management: Proceedings of the International Seminar and Workshop. Sabah, Malaysia, July 1997. Occasional paper of the IUCN Species Survival Commission No. 25.

Andrzejaczek, S., Schallert, R.J., Forsberg, K., Arnoldi, N.S., Cabanillas-Torpoco, M., Purizaca, W. and Block, B.A., 2021. Reverse diel vertical movements of oceanic manta rays off the northern coast of Peru and implications for conservation. Ecological Solutions and Evidence, 2(1), p.e12051.

Antonelis, G. A., J. D. Baker, T. C. Johanos, R. C. Braun, and A. L. Harting. 2006. Hawaiian monk seal (*Monachus schauinslandi*): Status and conservation issues. Atoll Research Bulletin 543:75-101.

Arendt, M., 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, Marine Resources Division.

Ari, C., and Correia, J. P. 2008. Role of sensory cues on food searching behavior of a captive *Manta birostris* (*Chondrichtyes, Mobulidae*). Zoo Biology. 27(4): 294-304. https://doi.org/10.1002/zoo.20189.

Baker, J., C. Littnan, and D. Johnston. 2006. Potential effecfs of sea-level rise on terrestrial habitat and biota of the northwestern Hawaiian Islands. Pages 3 *in* Twentieth Annual Meeting Society for Conservation Biology Conference, San Jose, California.

Balazs, G. H. 1982. Growth rates of immature green turtles in the Hawaiian Archipelago. Pages 117-125 *in* K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles. 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. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, NOAA-TM-NMFS-SWFC-36.

Balazs, G. H., S. G. Pooley, and S. K. Murakawa. 1995. Guidelines for handling marine turtles hooked or entangled in the Hawaii longline fishery: Results of an expert workshop held in Honolulu, Hawaii March 15-17,1995. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Honolulu.

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.

Beale, C., Stewart, J., Setyawan, E., Sianipar, A., Erdmann, M.V. 2019. Population dynamics of oceanic manta rays (Mobula birostris) in the Raja Ampat Archipelago, West Papua, Indonesia, and the impacts of the El Niño–Southern Oscillation on their movement ecology. Diversity and Distributions. 25. 10.1111/ddi.12962.

Bigelow, H. and Schroeder, W. 1953. In Sawfishes, Guitarfishes, Skates and Rays, Chimaeroids: Part 2 (pp. 1-2). New Haven: Yale University Press. https://doi.org/10.12987/9781933789255-003.

Bjorndal, K. A. 1982. The consequences of herbivory for life history pattern of the Caribbean green turtle, *Chelonia mydas*. Pages 111-116 *in* Biology and Conservation of Sea Turtles. Smithsonian Institution, Washington, D. C.

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

Bolten, A. B., K. A. Bjorndal, and H. R. Martins. 1994. Life history model for the loggerhead sea turtle (*Caretta caretta*) populations in the Atlantic: Potential impacts of a longline fishery. Pages 48-55 *in* G. J. Balazs, and S. G. Pooley, editors. Research Plan to Assess Marine Turtle Hooking Mortality, volume Technical Memorandum NMFS-SEFSC-201. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

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

Bouchard, S., and coauthors. 1998. Effects of exposed pilings on sea turtle nesting activity at Melbourne Beach, Florida. Journal of Coastal Research 14(4):1343-1347.

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., 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 58:12.

Braun R., J. Wester, C., Mcdonald, and J. H., Pate. 2024. Perceptions regarding the recreational cobia fishery in relation to manta ray conservation. Aquatic Conservervation: Marine and Freshwater Ecosystms. https://doi.org/10.1002/aqc.4147

Braun, C. D., G. B. Skomal, S. R. Thorrold, and M. L. Berumen. 2015. Movements of the reef manta ray (Manta alfredi) in the Red Sea using satellite and acoustic telemetry. Marine Biology 162(12):2351-2362.

Bresette, M., 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. Pages 288 *in* M. Frick, A. Panagopoulou, A. F. Rees, and K. Williams, editors. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

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. Royal Society Open Science 3(11):160717.

Caldwell, D. K., and A. Carr. 1957. Status of the sea turtle fishery in Florida. Pages 457-463 *in* J. B. Trefethen, editor Twenty-Second North American Wildlife Conference. Wildlife Management Institute, Statler Hotel, 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.

Carpentier, A.S., Berthe, C., Ender, I., Jaine, F.R., Mourier, J., Stevens, G., De Rosemont, M. and Clua, E., 2019. Preliminary insights into the population characteristics and distribution of reef (Mobula alfredi) and oceanic (M. birostris) manta rays in French Polynesia. Coral Reefs, 38(6), pp.1197-1210.

Carr, A. F. 1986. New perspectives on the pelagic stage of sea turtle development. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center.

Ceriani, S., A., P. Casale, M. Brost, E. H. Leone, and B. E. Witherington. 2019. Conservation implications of sea turtle nesting trends: elusive recovery of a globally important loggerhead population. ECOSPHERE Volume 10, Issue 11.

Chaloupka, M., and C. 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., 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.

Chaloupka, M. Y., and J. A. Musick. 1997a. 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, Florida.

Chaloupka, M. Y., and J. A. Musick. 1997b. 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.

CITES. 2013. Consideration of proposals for amendment of Appendices I and II: Manta Rays. Convention on International Trace in Endangered Species of Wild Fauna and Flora

(CITES), Sixteenth Meeting of the Conference of the Parties, CoP16 Prop. 46 (Rev. 2), Bangkok, Thailand.

Clark, T. B. 2010. Abundance, home range, and movement patterns of manta rays (Manta alfredi, M. birostris) in Hawai'i. Dissertation. Univeristy of Hawai'i at Mānoa, Honolulu, HI.

Colburn, T., D. Dumanoski, and J. P. Myers. 1996. Our stolen future. Dutton/ Penguin Books, New York.

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J., et al. (2013). Microplastic ingestion by zooplankton. Environ. Sci. Technol. 47, 6646–6655. doi: 10.1021/es400663f

Conant, T. A., and coauthors. 2009a. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Conant, T. A., and coauthors. 2009b. 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.

Convention on Migratory Species. 2014. Proposal for the inclusion of the reef manta ray (Manta alfredi) in CMS Appendix I and II. Convention on Migratory Species (CMS), 18th Meeting of the Scientic Council, UNEP/CMS/ScC18/Doc.7.2.9, Bonn, Germany.

Cook, M., and coauthors. 2016. Hooked on Kemp's - Preliminary results of Mississippi's angler survey. Pages 223-224 in L. Belskis, A. Frey, M. Jenson, R. LeRoux, and K. Stewart (compilers), editors. Proceedings of the Thirty-fourth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NOAA NMFS-SEFSC-701, Miami, FL.

Couturier, L. I. E., and coauthors. 2012. Biology, ecology and conservation of the Mobulidae. Journal of Fish Biology 80(5):1075-1119.

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.

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.

Daniels, R. C., T. W. White, and K. K. Chapman. 1993. Sea-level rise - destruction of threatened and endangered species habitat in South Carolina. Environmental Management 17(3):373-385.

Deakos, M. H., J. D. Baker, and L. Bejder. 2011. Characteristics of a manta ray Manta alfredi population off Maui, Hawaii, and implications for management. Marine Ecology Progress Series 429:245-260.

Dodd Jr., C. K. 1988. Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). U.S. Fish and Wildlife Service, 88(14).

Domínguez-Sánchez, P.S., Širović, A., Fonseca-Ponce, I.A. et al. Occupancy of acoustically tagged oceanic manta rays, Mobula birostris, in Bahia de Banderas, Mexico. Mar Biol 170, 128 (2023). https://doi.org/10.1007/s00227-023-04278-1

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

Dow, W., K. 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, Beaufort, North Carolina.

Duffy, L. and Griffiths, S. 2017. Resolving potential redundancy of productivity attributes to improve ecological risk assessments. SAC-08-07c. Inter-American Tropical Tuna Commission. Scientific Advisory Committee Eight Meeting, La Jolla, California (USA), 8-12 May 2017.

Dulvy, N. K., S. A. Pardo, C. A. Simpfendorfer, and J. K. Carlson. 2014. Diagnosing the dangerous demography of manta rays using life history theory. PeerJ Preprints 2.

DWH Trustees. 2015a. Deepwater Horizon Oil Spill: Draft Programmatic Damage Assessment and Restoration Plan and Draft Programmatic Environmental Impact Statement. Retrieved from http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan/.

DWH Trustees. 2015b. DWH Trustees (Deepwater Horizon Natural Resource Damage Assessment Trustees). 2015. Deepwater Horizon Oil Spill: Draft Programmatic Damage Assessment and Restoration Plan and Draft Programmatic Environmental Impact Statement. Retrieved from http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan/.

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(3):283-293.

Essumang, D.K. 2010. First determination of the levels of platinum group metals in Manta birostris (manta ray) caught along the Ghanaian coastline. Bulletin of Environmental Contamination and Toxicology 84: 720-725.

Farmer, N.A., Garrison, L.P., Horn, C. et al. (2022) The distribution of manta rays in the western North Atlantic Ocean off the eastern United States. Sci Rep 12, 6544. https://doi.org/10.1038/s41598-022-10482-8

Fish, M. R., and coauthors. 2005. Predicting the Impact of Sea-Level Rise on Caribbean Sea Turtle Nesting Habitat. Conservation Biology 19(2):482-491.

FitzSimmons, N. N., L. W. Farrington, M. J. McCann, C. J. Limpus, and C. Moritz. 2006. Green turtle populations in the Indo-Pacific: A (genetic) view from microsatellites. Pages 111 *in* N. Pilcher, editor Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation.

Foley, A. M., B. A. Schroeder, and S. L. MacPherson. 2008. Post-nesting migrations and resident areas of Florida loggerheads (Caretta caretta). Pages 75-76 *in* H. J. Kalb, A. S. Rhode, K. Gayheart, and K. Shanker, editors. Twenty-Fifth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Savannah, Georgia.

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., and coauthors. 2007. Characteristics of a green turtle (*Chelonia mydas*) assemblage in northwestern Florida determined during a hypothermic stunning event. Gulf of America Science 25(2):131-143.

Fossi, M. C., Baini, M., Panti, C., Galli, M., Jiménez, B., Muñoz-Arnanz, J., et al. (2017). Are whale sharks exposed to persistent organic pollutants and plastic pollution in the Gulf of California (Mexico)? first ecotoxicological investigation using skin biopsies. Comp. Biochem. Physiol. C Toxicol. Pharmacol. 199, 48–58. doi: 10.1016/j.cbpc.2017.03.002

Fossi, M. C., Marsili, L., Baini, M., Giannetti, M., Coppola, D., Guerranti, C., et al. (2016). Fin whales and microplastics: the Mediterranean Sea and the Sea of Cortez scenarios. Environ. Pollut. 209, 68–78. doi: 10.1016/j.envpol.2015.11.02

Fossi, M. C., Coppola, D., Baini, M., Giannetti, M., Guerranti, C., Marsili, L., et al. (2014). Large filter feeding marine organisms as indicators of microplastic in the pelagic environment: the case studies of the Mediterranean basking shark (Cetorhinus maximus) and fin whale (Balaenoptera physalus). Mar. Environ. Res. 100, 17–24. doi: 10.1016/j.marenvres.2014.02.002

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.

Freedman, R., Roy, S. S., Spatial patterning of Manta birostris in United States east coast offshore habitat, Applied Geography, Volume 32, Issue 2, 2012, Pages 652-659, ISSN 0143-6228, https://doi.org/10.1016/j.apgeog.2011.07.015. https://www.sciencedirect.com/science/article/pii/S0143622811001500.

Fretey, J. 2001. Biogeography and conservation of marine turtles of the Atlantic Coast of Africa, UNebraskaP/CMississippi Secretariat.

Galloway, T. S., and Lewis, C. N. (2016). Marine microplastics spell big problems for future generations. Proc. Nat. Acad. Sci. U.S.A. 113, 2331–2333. doi: 10.1073/pnas.1600715113

Garrett, C. 2004. Priority Substances of Interest in the Georgia Basin - Profiles and background information on current toxics issues. Canadian Toxics Work Group Puget Sound, Georgia Basin International Task Force, GBAP Publication No. EC/GB/04/79.

Garzon, F., Graham, R.T., Witt, M.J., Hawkes, L.A. 2020. Ecological niche modeling reveals manta ray distribution and conservation priority areas in Western Central Atlantic. Animal Conservation 24, 3 Pages 322-334

Garzon, F., Seymour, Z.T.A., Monteiro, Z.L. et al. Spatial ecology of a newly described oceanic manta ray population in the Atlantic Ocean. Mar Biol 170, 68 (2023). https://doi.org/10.1007/s00227-023-04219-y

Gavilan, F.M. 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. Marine Turtle Conservation in the Wider Caribbean Region - A Dialogue for Effective Regional Management, Santo Domingo, Dominican Republic.

Geraci, J. R. 1990. Physiologic and toxic effects on cetaceans. Pages 167-197 *in* J. R. Geraci, and D. J. S. Aubin, editors. Sea Mammals and Oil: Confronting the Risks. Academic Press, San Diego.

Germanov, E. S., Marshall, A. D., Bejder, L., Fossi, M. C., and Loneragan, N. R. (2018). Microplastics: no small problem for filter-feeding megafauna. Trends Ecol Evol. 33, 227–232. doi: 10.1016/j.tree.2018.01.005

Germanov, E. S., and A. D. Marshall. 2014. Running the gauntlet: regional movement patterns of Manta alfredi through a complex of parks and fisheries. PLOS ONE 9(10):e110071.

Germanov, E. S., and coauthors. 2019. Microplastics on the menu: Plastics pollute Indonesian manta ray and whale shark feeding grounds. Frontiers in Marine Science 6(679).

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

Gladys Porter Zoo. 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. 2016. (M. Barnette, personal communication, 2016).

Gladys Porter Zoo data. 2019 (M. Barnette, personal communication, 2019).

Graham, N. A. J., and coauthors. 2008. Climate Warming, Marine Protected Areas and the Ocean-Scale Integrity of Coral Reef Ecosystems. PLOS ONE 3(8):e3039.

Graham, R. T., and coauthors. 2012. Satellite tracking of manta rays highlights challenges to their conservation. PLOS ONE 7(5).

Grant, S. C. H., and P. S. Ross. 2002. Southern Resident killer whales at risk: Toxic chemicals in the British Columbia and Washington environment. Department of Fisheries and Oceans Canada, Sidney, B.C.

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

Groombridge, B. 1982. Kemp's ridley or Atlantic ridley, *Lepidochelys kempii* (Garman 1980). The IUCN Amphibia, Reptilia Red Data Book:201-208.

Guinder, V. A., and J. C. Molinero. 2013. Climate change effects on marine phytoplankton. Pages 68-90 in A. H. Arias, and M. C. Menendez, editors. Marine Ecology in a Changing World. CRC Press, Boca Raton, FL.

Guseman, J. L., and L. M. Ehrhart. 1992. Ecological geography of western Atlantic loggerheads and green turtles: Evidence from remote tag recoveries. Pages 50 *in* M. Salmon, and J. Wyneken, editors. Eleventh Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, Jekyll Island, Georgia.

Handwerk, B. 2010, October 16. Little-known Gulf Rays Affected by Oil Spill?. National Geographic News. https://www.nationalgeographic.com/animals/article/101015-new-manta-rays-gulf-bp-oil-spill-science-animals.

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 America: Opportunities for marine conservation. Biological Conservation 145:185-194.

Hartwell, S. I. 2004. Distribution of DDT in sediments off the central California coast. Marine Pollution Bulletin 49(4):299-305.

Harty K, Guerrero M, Knochel AM, Stevens GMW, Marshall A, Burgess K, Stewart JD. 2022. Demographics and dynamics of the world's largest known population of oceanic manta rays Mobula birostris in coastal Ecuador. Mar Ecol Prog Ser 700:145-159. https://doi.org/10.3354/meps14189

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 13:1-10.

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.

Hearn, A. R., and coauthors. 2014. Elasmobranchs of the Galapagos Marine Reserve. Pages 23-59 in J. Denkinger, and L. Vinueza, editors. Social and Ecological Interactions in the Galapagos Island, The Galapagos Marine Reserve: A dynamic social-ecological system. Springer, New York, NY.

Heinrichs, S., M. O'Malley, H. Medd, and P. Hilton. 2011. Global Threat to Manta and Mobula Rays. Manta Ray of Hope, 2011 Report.

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, D. T. Crouse, S. P. Epperly, and N. B. Frazer. 2003. Population models for Atlantic loggerheads: Past, present, and future. Pages 255-273 *in* A. Bolten, and B. Witherington, editors. Loggerhead Sea Turtles. Smithsonian Books, Washington, D. C.

Heppell, S. S., L. B. Crowder, and J. Priddy. 1995. Evaluation of a fisheries model for hawksbill sea turtle (Eretmochelys imbricata) harvest in Cuba. NOAA Tech. Memor. NMFS-OPR-5.

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

Herbst, L. H., and coauthors. 1995. An infectious etiology for green turtle fibropapillomatosis. Proceedings of the American Association for Cancer Research Annual Meeting 36:117.

Hilbourne S, Stevens G. 2019. Maldivian Manta Ray Project oceanic manta ray summary report 2019. The Manta Trust. https: //static1.squarespace.com/static/ 5a196500914e6b09132e911f/t/5d1090a94951180001880 a1d/1561366735894/MT_MMRP_Oceanic+Manta+ Summary+Report_2019_FINAL.pdf

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 22:105-112.

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

Hill, A. 2013. Rough Draft of Fishing Piers and Protected Species: An Asesment of the Presence and Effectiveness of Conservation Measures in Charlotte and Lee County, Florida. Pages 50 *in*. University of Miami, Rosenstiel School of Marine and Atmospheric Science.

Hirth, H. F. 1971. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus) 1758. Food and Agriculture Organization.

Hirth, H. F. 1997. Synopsis of the biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). Biological Report 91(1):120.

Holmberg Pierce, S.J. and Marshall, A.D., 2018. Photographic identification of sharks. Shark research: emerging technologies and applications for the field and laboratory, pp.219-234.

Hosegood, P.J., Nimmo-Smith, W.A.M., Proud, R., Adams, K. and Brierley, A.S., 2019. Internal lee waves and baroclinic bores over a tropical seamount shark ‘hot-spot’. Progress in Oceanography, 172, pp.34-50.

Iwata, H., S. Tanabe, N. Sakai, and R. Tatsukawa. 1993. Distribution of persistent organochlorines in the oceanic air and surface seawater and the role of ocean on their global transport and fate. Environmental Science and Technology 27(6):1080-1098.

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 Comparative Pathology 101:39-52.

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, volume NOAA-TM-NMFS-SWFSC-156.

Jakimska, A., Konieczka, P., Skóra, K., and Namieśnik, J. 2011. Bioaccumulation of metals in tissues of marine animals, part I: the role and impact of heavy metals on organism. Pol. J. Enviro. Studies 20, 1127–1146.

Johnson, S. A., and L. M. Ehrhart. 1994. Nest-site fidelity of the Florida green turtle. Pages 83 *in* B. A. Schroeder, and B. E. Witherington, editors. Thirteenth Annual Symposium on Sea Turtle Biology and Conservation.

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, G. P., M. I. McCormick, M. Srinivasan, and J. V. Eagle. 2004. Coral decline threatens fish biodiversity in marine reserves. Proc Natl Acad Sci U S A 101(21):8251-8253.

Kashiwagi, T., T. Ito, and F. Sato. 2010. Occurences of reef manta ray, Manta alfredi, and giant manta ray, M. birostris, in Japan, examined by photographic records. Japanese Society for Elasmobranch Studies 46:20-27.

Kashiwagi, T., A. D. Marshall, M. B. Bennett, and J. R. Ovenden. 2011. Habitat segregation and mosaic sympatry of the two species of manta ray in the Indian and Pacific Oceans: Manta alfredi and M. birostris. Marine Biodiversity Records 4:1-8.

Kitchen-Wheeler, A. M. (2010). Visual identification of individual manta ray (Manta alfredi) in the Maldives Islands, Western Indian Ocean . Marine Biology Research, 6(4), 351–363. https://doi.org/10.1080/17451000903233763.

Knochel, A.M., Cochran, J.E., Kattan, A., Stevens, G.M., Bojanowksi, E. and Berumen, M.L., 2022. Crowdsourced data reveal multinational connectivity, population demographics, and possible nursery ground of endangered oceanic manta rays in the Red Sea. Aquatic Conservation: Marine and Freshwater Ecosystems, 32(11), pp.1774-1786.

Kroetz, A.M., Mathers, A.N., Carlson, J.K., 2020. Evaluating protected species bycatch in the U.S. Southeast Gillnet Fishery. Fisheries Research, 228, https://doi.org/10.1016/j.fishres.2020.105573

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. Marine Turtle Conservation in the Wider Caribbean Region - A Dialogue for Effective Regional Management, Santo Domingo, Dominican Republic.

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

Lawson, J. M., and coauthors. 2017. Sympathy for the devil: a conservation strategy for devil and manta rays. PeerJ 5:e3027.

Lawson, J. M., and coauthors. 2016. Sympathy for the devil: A conservation strategy for devil and manta rays. PeerJ 5:e3027.

Lutcavage, M., P. Plotkin, B. Witherington, and P. Lutz. 1997. Human impacts on sea turtle survival. Pages 387–409 *in* P. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles, volume 1. CRC Press, Boca Raton, Florida.

Manta Trust. 2019a. Mooring Line Entanglement Mitigation. Manta Trust. Available online at: https://static1.squarespace.com/static/5a196500914e6b09132e911f/t/5dfe1e20b02a616945eb5772/1576934948296/MT_MMRP_Entanglement+Mitigation_2019_FINAL.pdf (accessed January 20, 2021).

Manta Trust. 2019b. Manta Ray Entanglement Protocol. Manta Trust. Available online at: https://static1.squarespace.com/static/5a196500914e6b09132e911f/t/5dfe2e70229b17607ddeb2c9/1576939123128/MT_MMRP_Entanglement+Protocol_2019_FINAL.pdf (accessed March 2, 2021).

Márquez M., R. 1990. Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date, Rome.

Márquez M., R. 1994. Synopsis of biological data on the Kemp's ridley sea turtle, *Lepidochelys kempii* (Garman, 1880). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Center.

Marshall, A., Barreto, R., Carlson, J., Fernando, D., Fordham, S., Francis, M.P., Derrick, D., Herman, K., Jabado, R.W., Liu, K.M., Rigby, C.L. & Romanov, E. 2022. Mobula birostris (amended version of 2020 assessment). The IUCN Red List of Threatened Species 2022: e.T198921A214397182. https://dx.doi.org/10.2305/IUCN.UK.2022-1.RLTS.T198921A214397182.en

Marshall, A. D., L. J. V. Compagno, and M. B. Bennett. 2009. Redescription of the genus Manta with resurrection of Manta alfredi (Krefft, 1868) (Chondrichthyes; Myliobatoidei; Mobulidae). Zootaxa 2301:1-28.

Marshall, A. D., and M. B. Bennett. 2010. Reproductive ecology of the reef manta ray *Manta alfredi* in southern Mozambique. Journal of Fish Biology 77(1): 169-190.

Matkin, C. O., and E. Saulitis. 1997. Restoration notebook: Killer whale (*Orcinus orca*). *Exxon Valdez* Oil Spill Trustee Council, Anchorage, Alaska.

McGregor, F., Richardson, A. J., Armstrong, A. J., Armstrong, A. O., and C. L. Dudgeon. 2019. Rapid wound healing in a reef manta ray masks the extent of vessel strike. https://doi.org/10.1371/journal.pone.0225681.

McMichael, E., R. R. Carthy, and J. A. Seminoff. 2003. Evidence of homing behavior in juvenile green turtles in the northeastern Gulf of America. Pages 223-224 *in* J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation.

Medeiros, A. M., O. J. Luiz, and C. Domit. 2015. Occurrence and use of an estuarine habitat by giant manta ray Manta birostris. Journal of Fish Biology 86(6):1830-1838.

Meylan, A., B. Schroeder, and A. Mosier. 1994. Marine turtle nesting activity in the State of Florida, 1979-1992. Pages 83 *in* K. A. Bjorndal, A. B. Bolten, D. A. Johnson, and P. J. Eliazar, editors. Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.

Meylan, A. B., B. A. Schroeder, and A. Mosier. 1995. Sea turtle nesting activity in the State of Florida 1979-1992. Florida Department of Environmental Protection (52):63.

Milessi, A. C., and M. C. Oddone. 2003. Primer registro de Manta birostris (Donndorff 1798) (Batoidea: Mobulidae) en el Rio de La Plata, Uruguay. Gayana 67(1):126-129.

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, volume II. CRC Press, Boca Raton, Florida.

Moncada, F., A. Abreu-Grobois, D. Bagley, K. A. Bjorndal, A. B. Bolten, J. A. Caminas, L. Ehrhart, A. Muhlia-Melo, G. Nodarse, B. A. Schroeder, J. Zurita, and L. A. Hawkes. 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.

Moore, A. B. M. 2012. Records of poorly known batoid fishes from the north-western Indian Ocean (Chondrichthyes: Rhynchobatidae, Rhinobatidae, Dasyatidae, Mobulidae). African Journal of Marine Science 34(2):297-301.

Mourier, J. 2012. Manta rays in the Marquesas Islands: First records of Manta birostris in French Polynesia and most easterly location of Manta alfredi in the Pacific Ocean, with notes on their distribution. Journal of Fish Biology 81(6):2053-2058.

Murphy, T. M., and S. R. Hopkins. 1984. Aerial and ground surveys of marine turtle nesting beaches in the southeast region. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center.

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, New York, New York.

NMFS-NEFSC. 2011. Preliminary summer 2010 regional abundance estimate of loggerhead turtles (*Caretta caretta*) in northwestern Atlantic Ocean continental shelf waters. U.S. Department of Commerce, Northeast Fisheries Science Center, Reference Document 11-03.

NMFS-SEFSC. 2009a. An assessment of loggerhead sea turtles to estimate impacts of mortality on population dynamics. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, PRD-08/09-14.

NMFS-SEFSC. 2009b. An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. NMFS Southeast Fisheries Science Center.

NMFS. 1997. Endangered Species Act Section 7 Consultation - Biological Opinion on Navy activities off the southeastern United States along the Atlantic coast, National Marine Fisheries Service, Office of Protected Resources and the Southeast Regional Office.

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. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

NMFS. 2012. Protocols for Categorizing Sea Turtles for Post-release Mortality Estimates. August 2001, revised February 2012. PRD Contribution: #PRD-2011-07. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, Florida.

NMFS. 2019. Giant manta ray recovery outline. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS. 2020. Endangered Species Act Section 7 Biological Opinion on the Pelagic Longline Fishery for Atlantic Highly Migratory Species. F/SER/2014/00006[13697]

NMFS. 2021. Endangered Species Act Section 7 Biological Opinion on the Reinitiation of Endangered Species Act Section 7 Consultation on the Implementation of the Sea Turtle Conservation Regulations under the ESA and the Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act. SERO-2021-00087

NMFS. 2023. Endangered Species Act Section 7 Biological Opinion on the Amendment to the 2015 Biological Opinion on the operation of the Coastal Migratory Pelagic Fishery as Managed by the Fishery Management Plan for Coastal Migratory Pelagic Resources in the Gulf of America and Atlantic Region under the Magnuson-Stevens Fishery Conservation and Management Act. SERO-2021-03456.

NMFS. 2024. Endangered Species Act recovery status review for giant manta rays (mobula birostris). U.S. Department of Commerce, National Oceanic and Atmospheric Adminstartion, National Marine Fisheries Service, Office of Protected Resources, Silver Springs MD.

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

NMFS, and USFWS. 1992. Recovery plan for leatherback turtles *Dermochelys coriacea* in the U. S. Carribean, Atlantic and Gulf of America. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 1993. Recovery plan for the hawksbill turtle *Eretmochelys imbricata* in the U.S. Caribbean, Atlantic and Gulf of America. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, St. Petersburg, Florida.

NMFS, and USFWS. 2007a. Green Sea Turtle (*Chelonia mydas*) 5-year review: Summary and Evaluation. National Marine Fisheries Service, Silver Spring, Maryland.

NMFS, and USFWS. 2007b. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2007c. Loggerhead sea turtle (*Caretta caretta*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2008. Recovery plan for the northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*), second revision. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS, USFWS, and SEMARNAT. 2011. Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. National Marine Fisheries Service, Silver Spring, Maryland.

NMFS and USFWS. 1991b. 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, Office of Protected Resources, Silver Spring, Maryland.

NMFS and USFWS. 2007d. Green Sea Turtle (*Chelonia mydas*) 5-year review: Summary and Evaluation. National Marine Fisheries Service, Silver Spring, Maryland.

NMFS and USFWS. 2009. Recovery plan for the northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta)*. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

Notarbartolo di Sciara, G., and E. V. Hillyer. 1989. Mobulid rays off eastern Venezuela (Chondrichthyes, Mobulidae). Copeia (3):607-614.

NRC. 1990. Decline of the sea turtles: Causes and prevention. National Research Council, Washington, D. C.

Ogren, L. H. 1989. Distribution of juvenile and subadult Kemp's ridley sea turtles: Preliminary results from 1984-1987 surveys. Pages 116-123 *in* C. W. Caillouet Jr., and A. M. Landry Jr., editors. First International Symposium on Kemp's Ridley Sea Turtle Biology, Conservation and Management. Texas A&M University, Sea Grant College, Galveston, Texas.

Ooi, M.S.M., Townsend, K.A., Bennett, M.B., Richardson, A.J., Fernando, D., Villa, C.A. and Gaus, C. 2015. Levels of arsenic, cadmium, lead and mercury in the branchial plate and muscle tissue of mobulid rays. Marine Pollution Bulletin 94(1): 251-259.

Pate, J. Evidence of reproductive and feeding habitat for manta rays off Florida's Atlantic coast. Environ Biol Fish (2024). https://doi.org/10.1007/s10641-024-01566-7

Pate, J. H., and Marshall, A. D. 2020. Urban manta rays: potential manta ray nursery habitat along a highly developed Florida coastline. Endanger. Species Res. 43, 51–64. doi: 10.3354/esr01054

Pate J.H, Macdonald C., Wester J. 2020. Surveys of recreational anglers reveal knowledge gaps and positive attitudes towards manta ray conservation in Florida. Aquatic Conservation Marine and Freshwater Ecosystems. 31. 10.1002/aqc.3508.

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

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

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

Rambahiniarison, J. M., and coauthors. 2018. Life history, growth, and reproductive biology of four mobulid species in the Bohol Sea, Philippines. Frontiers in Marine Science 5:269.

Rebel, T. P. 1974. Sea Turtles and the Turtle Industry of the West Indies, Florida and the Gulf of America. University of Miami Press, Coral Gables, Florida.

Restrepo, J., E.G. Webster, I. Ramos, R.A. Valverde. 2023. Recent decline of green turtle Chelonia mydas nesting trends at Tortuguero, Costa Rica. Endangered Species Research 51: 59-72.

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.

Roberts, B. 2022. Catch Cobia on migrating manta rays: cobia can't resist hanging around these giant rays on the move this month. Florida Sportsman. First published Florida Sportsman April 2015. https://www.floridasportsman.com/editorial/cobia-rays/400722

Rochman, C. M., Kurobe, T., Flores, I., and Teh, S. J. (2014). Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Sci. Total Environ. 493, 656–661. doi: 10.1016/j.scitotenv.2014.06.051

Rohner, C.A., Flam, A.L., Pierce, S.J. and Marshall, A.D. 2017. Steep declines in sightings of manta rays and devilrays (Mobulidae) in southern Mozambique. PeerJ Preprints 5: e3051v1.

Romo-Curiel, A.E., Z. Ramírez-Mendoza, A. Fajardo-Yamamoto, M.R. Ramírez-León, M.C. García-Aguilar, S.Z. Herzka, P. Pérez-Brunius, L.E. Saldaña-Ruiz, J. Sheinbaum, K. Kotzakoulakis, J. Rodríguez-Outerelo, F. Medrano, O. Sosa-Nishizaki. 2022. Assessing the exposure risk of large pelagic fish to oil spills scenarios in the deep waters of the Gulf of America, Marine Pollution Bulletin, Volume 176, 113434, https://doi.org/10.1016/j.marpolbul.2022.113434.

Ryder, C. E., T. A. Conant, and B. A. Schroeder. 2006. Report of the Workshop on Marine Turtle Longline Post-Interaction Mortality. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

Schmid, J. R., and J. A. Barichivich. 2006. *Lepidochelys kempii*–Kemp's ridley. Pages 128-141 *in* P. A. Meylan, editor. Biology and conservation of Florida turtles. Chelonian Research Monographs, volume 3.

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. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, Florida.

Schroeder, B. A., and A. M. Foley. 1995. Population studies of marine turtles in Florida Bay. J. I. Richardson, and T. H. Richardson, editors. Twelfth Annual Workshop on Sea Turtle Biology and Conservation.

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

Seminoff, J. A., and coauthors. 2015. Status review of the green turtle (*Chelonia Mydas*) under the endangered species act. NOAA Technical Memorandum, NMFS-SWFSC-539.

Setälä, O., Fleming-Lehtinen, V., and Lehtiniemi, M. 2014. Ingestion and transfer of microplastics in the planktonic food web. Environ. Pollut. 185, 77–83. doi: 10.1016/j.envpol.2013.10.013

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.

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

Stewart, J.D., Jaine, F.R.A., Armstrong, A.J., Armstrong, A.O., Bennett, M.B., Burgess, K.B., Couturier, L.I.E., Croll, D.A., Cronin, M.R., Deakos, M.H., Dudgeon, C.L., Fernando, D., Froman, N., Germanov, E.S., Hall, M.A., Hinojosa-Alvarez, S., Hosegood, J.E., Kashiwagi, T., Laglbauer, B.J.L., Lezama-Ochoa, N., Marshall, A.D., McGregor, F., Notarbartolo di Sciara, G., Palacios, M.D., Peel, L.R., Richardson, A.J., Rubin, R.D., Townsend, K.A., Venables, S.K. and Stevens, G.M.W. 2018a. Research priorities to support effective manta and devil ray conservation. Frontiers in Marine Science 5(314).

Stewart, J. D., Barroso, A., Butler, R. H., and Munns, R. J. 2018b. Caught at the surface: myctophids make easy prey for dolphins and devil rays. Ecology. 99, 1894–1896. doi: 10.1002/ecy.2348

Stewart, J. D., Beale, C. S., Fernando, D., Sianipar, A. B., Burton, R. S., Semmens, B. X., et al. (2016a). Spatial ecology and conservation of manta birostris in the indo-pacific. Biol. Conserv. 200, 178–183. doi: 10.1016/j.biocon.2016.05.016

Stewart, J. D., Hoyos-Padilla, E. M., Kumli, K. R., and Rubin, R. D. 2016b. Deep-water feeding and behavioral plasticity in Manta birostris revealed by archival tags and submersible observations. Zoology 119, 406–413. doi: 10.1016/j.zool.2016.05.010.

Stewart JD, Rohner CA, Araujo G, Avila J and others. 2017. Trophic overlap in mobulid rays: insights from stable isotope analysis. Mar Ecol Prog Ser 580:131-151. https://doi.org/10.3354/meps12304.

Stock, C. A., Dunne, J. P., and J. G. John. 2014. Drivers of trophic amplification of ocean productivity trends in a changing climate, Biogeosciences, 11, 7125–7135, https://doi.org/10.5194/bg-11-7125-2014, 2014.

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.

Strike, E. M., Harris, J. L., Ballard, K. L., Hawkins, J. P., Crockett, J. C., and G. M. W., Stevens. 2022. Sublethal Injuries and Physical Abnormalities in Maldives Manta Rays, *Mobula alfredi* and *Mobula birostris*. Marine Conservation and Sustainability. Vol 9.

TEWG. 1998a. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the Western North Atlantic. Department of Commerce, Turtle Expert Working Group.

TEWG. 1998b. An assessment of the Kemp’s ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the western North Atlantic. U. S. Dept. Commerce.

TEWG. 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Turtle Expert Working Group.

TEWG. 2009. An assessment of the loggerhead turtle population in the western North Atlantic ocean. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Turtle Expert Working Group, NMFS-SEFSC-575.

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. 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.

USFWS and NMFS. 1998. Endangered Species Act consultation handbook. Procedures for Conducting Section 7 Consultations and Conferences. U.S. Fish and Wildlife, National Marine Fisheries Service.

Vargo, S., P. Lutz, D. Odell, E. V. Vleet, and G. Bossart. 1986. Study of the effects of oil on marine turtles. U.S. Department of the Interior, Minerals Management Service, Vienna, Virginia.

Ward-Paige, C.A., David, B. and Worm, B. 2013. Global population trends and human use patterns of Manta and Mobula rays. PLoS ONE 8(9): e74835. doi:10.1371/journal.pone.0074835.

Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster. 2005. Fishing methods to reduce sea turtle mortality associated with pelagic longlines. Canadian Journal of Fisheries and Aquatic Sciences 62(5):965-981.

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.

Wershoven, J. L., and R. W. Wershoven. 1992. Juvenile green turtles in their nearshore habitat of Broward County, Florida: A five year review. Pages 121-123 *in* M. Salmon, and J. Wyneken, editors. Eleventh Annual Workshop on Sea Turtle Biology and Conservation.

White, E.R., Myers, M.C., Flemming, J.M. and Baum, J.K. 2015. Shifting elasmobranch community assemblage at Cocos Island--an isolated marine protected area. Conservation Biology 29(4): 1186–1197

White, F. N. 1994. Swallowing dynamics of sea turtles. Pages 89-95 *in* G. H. Balazs, and S. G. Pooley, editors. Research Plan to Assess Marine Turtle Hooking Mortality. National Oceanic and Atmospheric Administration, Honolulu, Hawaii.

Witherington, B., M. Bresette, and R. Herren. 2006. *Chelonia mydas* - Green turtle. Chelonian Research Monographs 3:90-104.

Witherington, B., S. Hirama, and A. Moiser. 2003. Effects of beach armoring structures on marine turtle nesting. U.S. Fish and Wildlife Service.

Witherington, B., S. Hirama, and A. Moiser. 2007. Changes to armoring and other barriers to sea turtle nesting following severe hurricanes striking Florida beaches. U.S. Fish and Wildlife Service.

Witherington, B. E. 1992. Behavioral responses of nesting sea turtles to artificial lighting. Herpetologica 48(1):31-39.

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., and K. A. Bjorndal. 1991. Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. Biological Conservation 55(2):139-149.

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

Witherington, B. E., and L. M. Ehrhart. 1989b. Status, and reproductive characteristics of green turtles (*Chelonia mydas*) nesting in Florida. Pages 351-352 *in* L. Ogren, and coeditors, editors. Second Western Atlantic Turtle Symposium. .

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

Work, T. M. 2000. Synopsis of necropsy findings of sea turtles caught by the Hawaii-based pelagic longline fishery.

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.

Zurita, J. C., and coauthors. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. Pages 25-127 *in* J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation, Miami, Florida.

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 Maryland Herpetological Society 13(3):170-192.