SPOT Record of Decision

APPENDIX I

Endangered Species Act - Section 7 Consultation Biological Opinion

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Southeast Regional Office
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11/09/2022

F/SER31:DPO/MT
SERO-2020-00075
https://doi.org/10.25923/gx4d-xz46

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Ref.: MARAD-2019-0011, SPOT Terminal Services LLC, and MARAD-2019-0093, Texas GulfLink Holdings Deepwater oil exportation ports construction, operation, and decommissioning, Continental Shelf in Galveston/Freeport area, Brazoria County, Texas

Dear Lt. Commander Matthew Meacham and Yvette Fields,

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

Permit Number(s)	Applicant(s)	SER Number	Project Type(s)
MARAD-2019-0011	SPOT Terminal Services LLC	SERO-2020-00075	Deepwater oil exportation port construction/ operation/ decommissioning
MARAD-2019-0093	GulfLink LLC	SERO-2020-03309	Deepwater oil exportation port construction/ operation/ decommissioning

This Opinion considers the effects of constructing, operating, and decommissioning two deepwater crude oil exportation ports by the project applicants on the following listed species and critical habitat: sperm whale, Rice's whale, Green (North Atlantic and South Atlantic

Distinct Population Segments [DPS]), Kemp's ridley, leatherback, loggerhead (Northwest Atlantic DPS), and hawksbill sea turtles, giant manta ray, oceanic whitetip shark, elkhorn, boulder star, mountainous star, and lobed star coral, and loggerhead Northwest DPS critical habitat. NMFS concludes that the proposed actions are not likely to adversely affect Rice's whale and ESA-listed corals, and are likely to adversely affect, but are not likely to jeopardize the continued existence of sperm whale, green, Kemp's ridley, leatherback, loggerhead, and hawksbill turtles; giant manta ray, and oceanic whitetip shark. NMFS also concludes that the proposed actions are likely to adversely affect, but are not likely to result in the destruction or adverse modification of designated critical habitat for loggerhead sea turtle.

NMFS is providing an Incidental Take Statement (ITS) with the Opinion. The ITS describes reasonable and prudent measures NMFS considers necessary or appropriate to minimize the impact of incidental take associated with these actions. The ITS also specifies terms and conditions, including monitoring and reporting requirements with which the United States Coast Guard, Maritime Administration, SPOT Terminal Services LLC, and Texas GulfLink Holdings must comply to carry out the reasonable and prudent measures.

We look forward to further cooperation with you on other projects to ensure the conservation of our threatened and endangered marine species and designated critical habitat. If you have any questions on this consultation, please contact Daniel Owen, Consultation Biologist, by phone at 727-209-5961, or by email at Daniel.Owen@noaa.gov, and Michael Tucker, Consultation Biologist, at (727) 209-5981, or by email at michael.tucker@noaa.gov.

Sincerely,

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Regional Administrator

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Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	United States Coast Guard and Maritime Administration
Applicant:	SPOT Terminal Services LLC and Texas GulfLink Holdings
	MARAD-2019-0011; MARAD-2019-0093
Activity:	Deepwater oil exportation port construction, operation, and decommissioning, Continental Shelf in Galveston/Freeport area, Brazoria Counties, Texas
Consulting Agency:	National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida
	Tracking Number SERO-2020-00075; SERO-2020-03309 https://doi.org/10.25923/gx4d-xz46
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	Andrew J. Strelcheck, Regional Administrator NMFS, Southeast Regional Office St. Petersburg, Florida
Date Issued:	11/09/2022

TABLE OF CONTENTS

TABLE OF CONTENTS 4
LIST OF FIGURES 4
LIST OF TABLES 6
ACRONYMS AND ABBREVIATIONS 7
UNITS OF MEASUREMENT 10
INTRODUCTION 11
1 CONSULTATION HISTORY 11
2 DESCRIPTION OF THE PROPOSED ACTIONS AND ACTION AREA 12
3 STATUS OF LISTED SPECIES AND CRITICAL HABITAT 47
4 ENVIRONMENTAL BASELINE 122
5 EFFECTS OF THE ACTION ON SPECIES AND CRITICAL HABITAT 143
6 CUMULATIVE EFFECTS 160
7 INTEGRATION AND SYNTHESIS 161
8 CONCLUSION 192
9 INCIDENTAL TAKE STATEMENT 193
10 CONSERVATION RECOMMENDATIONS 197
11 REINITIATION OF CONSULTATION 199
12 LITERATURE CITED 200

LIST OF FIGURES

Figure 1. SPOT DWP and components diagram (Figure 2.2-4 in the Sea Port Oil Terminal Deepwater Port Project Final Environmental Impact Statement [SPOT FEIS], July 2022) 14
Figure 2. Relative Ship and Channel Sizes (Figure 1.1-4 in the SPOT FEIS, July 2022) 15
Figure 3. SPOT DWP Lease blocks and pipeline route (Figure 2.2-3 in the SPOT FEIS, July 2022) 16
Figure 4. SPOT DWP Pile Jacket System (Figure 2.2-6 in the SPOT FEIS, July 2022) 20
Figure 5. An unconfined, single-ring bubble curtain system for the 30-in PLEM piles (Figure 341-2 in the SPOT Deepwater Port License Application Data Gap Response #14 – Part A, April 2021) 22
Figure 6. Image of an operating, weighted, unconfined, single-ring bubble curtain system for the 30-in PLEM piles (Figure 341-3 in the SPOT Deepwater Port License Application Data Gap Response #14 – Part A, April 2021) 23
Figure 7. SPM Buoy Mooring Schematic (Figure 2.2-7 in the SPOT FEIS, July 2022) 25
Figure 8. Overview of the proposed TGL DWP area and facilities (Figure 3-1 in the TGL DWP Biological Assessment, November 2020) 33
Figure 9 Schematic drawing of proposed TGL offshore platform (Figure 2.2-5 in the TGL DEIS, November 2020) 37
Figure 10. Schematic drawing of proposed bubble curtain design, plan view (drawing provided by Abadie Engineering LLC, April 2021) 39
Figure 11. Schematic drawing of proposed bubble curtain design, side view (drawing provided by Abadie Engineering LLC, April 2021) 39
Figure 12. The SPOT deep water port location site (NAD83 is 28.466394, -95.123473) and TGL deep water port location (NAD83 is 28.55167, -95.02833) (©2022 Google) 45
Figure 13. Worst-case impact area for SPOT DWP (Figure 4.6-13 in the SPOT FEIS, July 2022) 46

Figure 14. Worst-case impact area for TGL DWP (Figure 3-63 in Appendix I of TGL DEIS, October 2020) 46
Figure 15. Likely shipping lanes for carriers utilizing the DWPs (NMFS stock figure, updated May 2022) 46
Figure 16. Distribution of Rice's Whale based on sightings (yellow "core area") and sightings coupled with passive acoustic monitoring (green "extended habitat). Adapted from Farmer et al. (2022). 54
Figure 17. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific 64
Figure 18. Green sea turtle nesting at Florida index beaches since 1989 69
Figure 19. Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019 and CONAMP data 2020, 2021) 74
Figure 20. Leatherback sea turtle nesting at Florida index beaches since 1989 82
Figure 21. Loggerhead sea turtle nesting at Florida index beaches since 1989 89
Figure 22. South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: https://www.dnr.sc.gov/seaturtle/ibs.htm) 90
Figure 23. The Extent of Occurrence (dark blue) and Area of Occupancy (light blue) based on species distribution (Lawson et al. 2017). 99
Figure 24. Geographic distribution of oceanic whitetip shark (Last and Stevens 2009) 106
Figure 25. Recapture distribution for the oceanic whitetip shark from the NMFS Co-Operative Shark Tagging Program during 1962-1993 and NMFS Unpublished Data. 107
Figure 26. Map with bottom depth showing filtered tracks for nine oceanic whitetip sharks equipped with Standard Rate tags. Colored lines represent tracks from individuals (listed by tag ID) (Howey-Jorden et al. 2013) 109
Figure 27. Sperm whale sightings and predicted densities in the Gulf of Mexico 113
Figure 28. Distribution of critical habitat for the NWA DPS of loggerhead sea turtles 119
Figure 29. *Sargassum* critical habitat 121
Figure 30. Predicted average contribution to ambient sound from modeled sound sources including seismic airgun surveys at different depths for 50 Hz and 100 Hz. Source: Marine Geospatial Ecology Lab, Duke University (2012) as published on CetSound website. 133
Figure 31. Five HARP locations, which collected data over several months during 2010-2013, are displayed as squares notated with site codes [GC: Green Canyon; MC: Mississippi Canyon; MP: Main Pass; DC: De Soto Canyon; and DT: Dry Tortugas]. The triangle is a NOAA weather bouy station used to measure wind speeds. Figure from (Wiggins et al. 2016). 134
Figure 32. Figure from Berenshtein et al. (2020a) showing spatiotemporal dynamics of the spill for dates showing cumulative oil concentrations in figures G- 15 May 2010; J- 18 June 2010; and M- 2 July 2010 136
Figure 33. Oil Fate and Trajectory Modeling Overview (Figure 4.6-7 in the SPOT FEIS, July 2022) 146
Figure 34. Selected Spill Locations: Nearshore and at the Deepwater Port (Figure 4.6-8 in the SPOT FEIS, July 2022) 147
Figure 35. Locations of release points for Scenario 1 & 3 (SPOT DWP) and Scenario 2 (2 miles from shore) near Freeport Texas (Figure 4.6-9 in the SPOT FEIS, July 2022) 149
Figure 36. Potential Oil Spill Effect on Shoreline from a Double Pipeline Breach 2 Miles from Shore (Scenario 2, Figure 4.6-10 in the SPOT FEIS, July 2022) 150

Figure 37. Potential Oil Spill Effect on Shoreline from a Double Pipeline Breach at the SPOT DWP (Scenario 1, Figure 4.6-11 in the SPOT FEIS, July 2022).. 151
Figure 38. Potential Oil Spill Effect on Shoreline from a Full Release of Cargo from a VLCC at the SPOT DWP (Scenario 3, Figure 4.6-12 in the SPOT FEIS, July 2022)............................ 152
Figure 39. 2 Miles from Shoreline Scenario (Scenario 2) of Gulf of Mexico Potential Oil Spill Impact (Figure 4.6-13 in the SPOT FEIS, July 2022) 153
Figure 40. Pipeline Breach at SPOT DWP Scenario (Scenario 1) of Gulf of Mexico Potential Oil Spill Impact (Figure 4.6-14 in the SPOT FEIS, July 2022).. 154
Figure 41. VLCC Scenario (Scenario 3) of Gulf of Mexico Potential Oil Spill Impact (Figure 4.6-15 in the SPOT FEIS, July 2022) 154

LIST OF TABLES

Table 1. Pile-Driving Details for the 2 Types of Structures 21
Table 2. Time Period for Offshore Construction Activities 28
Table 3. Pile-Driving Details for Each of the 3 Types of Structures.. 38
Table 4. Option 1 and 2 Decommissioning Methods for Offshore Components 43
Table 5. Effects Determination(s) for Species the Action Agency Agencies and/or NMFS Believe May Be Affected by the Proposed Actions 48
Table 6. Effects Determination(s) for Designated Critical Habitat the Action Agencies and NMFS Believe May Be Affected by the Proposed Actions 49
Table 7. Number of Leatherback Sea Turtle Nests in Florida 82
Table 8. Total Number of NRU Loggerhead Nests (GADNR, SCDNR, and NCWRC nesting datasets compiled at Seaturtle.org) 90
Table 9. Description of critical habitat for the NWA DPS of loggerhead sea turtles................ 119
Table 10. Likelihood of Occurrence for Oil Spills over 1,000 bbl from Platforms, Pipelines, and Tankers in the BOEM GoM Western Planning Area (WPA) (2012-2051) (NMFS, 2020) 135
Table 11. Likelihood of Occurrence for Oil Spills over 10,000 bbl from Platforms, Pipelines, and Tankers in the BOEM GoM (WPA) (2012-2051) (NMFS, 2020).. 135
Table 12. Average Number and Size of Spills projected by BOEM to Occur on the GoM OCS Resulting from Permitted Lease Actions on Leases Awarded through 2027 (NMFS, 2020)..... 135
Table 13. Oceanic Juvenile Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015.. 137
Table 14. Large Juveniles and Adult Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015).. 137
Table 15. Oceanic Juvenile Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015) 155
Table 16. Large Juveniles and Adult Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015).. 155
Table 17. Estimates of Oceanic Juvenile Sea Turtles that May Be Exposed and Killed by a Potential Worst Credible Oil Spill Discharge from the Proposed Projects Scaled to 14% of the Effects of the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)... 156
Table 18. Estimates of Large Juveniles and Adult Sea Turtles that May be Exposed and Killed by a Potential Worst Credible Oil Spill Discharge from the Proposed Projects Scaled to 14% of the Effects of the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)... 156
Table 19. Summary of the Effects of a Worst Credible Oil Spill Discharge on NA DPS of Green Sea Turtles 166

Table 20. Summary of the Effects of a Worst Credible Oil Spill Discharge on SA DPS of Green Sea Turtles 169
Table 21. Summary of the Effects of a Worst Credible Oil Spill Discharge on Kemp's Ridley Sea Turtles 172
Table 22. Summary of the Effects of a Worst Credible Oil Spill Discharge on NWA DPS of Loggerhead Sea Turtles 175
Table 23. Summary of the Effects of a Worst Credible Oil Spill Discharge on Hawksbill Sea Turtles 179
Table 24. Summary of the Effects of a Worst Credible Oil Spill on Leatherback Sea Turtles .. 182
Table 25. Summary of Integration and Synthesis for ESA-Listed Species Likely to Be Adversely Affected........ 189
Table 26. Summary of Integration and Synthesis for Designated Critical Habitat based on review of effects to PBFs from oil spills. 192

ACRONYMS AND ABBREVIATIONS

ASME	American Society of Mechanical Engineers
BIRNM	Buck Island Reef National Monument
BOEM	Bureau of Ocean Energy Management
BSEE	Bureau of Safety and Environmental Enforcement
CCL	curved carapace length
CFR	Code of Federal Regulations
CITES	Convention on International Trade in Endangered Species
CMP	Coastal Migratory Pelagics
CPUE	catch per unit effort
CSTP	Cooperative Shark Tagging Program
DDT	dichlorodiphenyltrichloroethane
DEIS	Draft Environmental Impact Statement
DNA	deoxyribonucleic acid
DPI	Direct Pipe Installation
DTRU	Dry Tortugas Recovery Unit
DWH	Deepwater Horizon
DWP	Deepwater port
DWPA	Deepwater Port Act of 1974
DWPSP	Deepwater Port Security Plan
DPS	Distinct Population Segment
EEZ	Exclusive Economic Zone
EFGB	East Flower Garden Bank
EFH	Essential Fish Habitat
EPO	Enterprise Products Operating LLC
ESA	Endangered Species Act
FEIS	Final Environmental Impact Statement
FGBNMS	Flower Garden Bank National Marine Sanctuary
FMP	Fisheries Management Plan
FP	Fibropapillomatosis
FR	Federal Register
FRP	Facility Response Plan

FSP	Facility Security Plan
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
GADNR	Georgia Department of Natural Resources
GCRU	Greater Caribbean Recovery Unit
GIWW	Gulf Intracoastal Waterway
GoM	Gulf of Mexico
GRBO	Gulf of Mexico Regional Biological Opinion
HARP	high-frequency acoustic recording package
HDD	Horizontal Direction Drill
HMS	Highly Migratory Species
ICWW	Intracoastal Waterways
IMO	International Maritime Organization
IPCC	Intergovernmental Panel on Climate Change
ITS	Incidental Take Statement
IUU	illegal, unreported, and unregulated
IWC	International Whaling Commission
LLC	Limited Liability Company
MARAD	Maritime Administration
MARPOL	Prevention of Pollution of Ships
MHW	Mean High Water
MLLW	Mean Lower Low Water
MMPA	Marine Mammal Protection Act
MPA	Marine Protected Areas
MSA	Magnuson-Stevens Fishery Conservation and Management Act
mtDNA	mitochondrial DNA
NMFS	National Marine Fisheries Service
NA	North Atlantic
NCWRC	North Carolina Wildlife Resources Commission
NGMRU	Northern Gulf of Mexico Recovery Unit
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NRU	Northern Recovery Unit
NTL	Notice to Lessees and Operators
NWA	Northwest Atlantic
OCS	Outer Continental Shelf
Opinion	Biological Opinion
OPR	Office of Protected Resources
OPSMAN	Port Operations Manual
OSV	offshore supports vessel
PAM	passive acoustic monitoring
PCB	polychlorinated biphenyls
PCE	primary constituent element
PDARP	Programmatic Damage Assessment and Response Plan
PFC	perfluorinated chemicals
PFRU	Peninsular Florida Recovery Unit

PHMSA	Pipeline and Hazardous Safety Administration
PIANC	World Association for Waterborne Transport Infrastructure
PK	peak sound pressure level
PLEM	pipeline end manifold
PLL	pelagic longline
PMMP	Prevention, Monitoring, and Maintenance Plan
PRDNER	Puerto Rico Department of Natural and Environmental Resources
PSO	Protected Species Observer
RCP	representative concentration pathways
RMS	root-mean-square pressure level
ROV	remotely operated vehicle
ROW	right-of-way
RPM	Reasonable and Prudent Measure
SA	South Atlantic
SAV	Submerged Aquatic Vegetation
SCDNR	South Carolina Department of Natural Resources
SCL	straight carapace length
SDEIS	Supplemental Draft Environmental Impact Statement
SEFSC	Southeast Fisheries Science Center
SEL	sound exposure level
SELcum	cumulative sound exposure level
SERO	Southeast Regional Office
SPM	single point mooring
SPOT	Sea Port Oil Terminal
SPR	Strategic Petroleum Reserve
SST	sea surface temperature
TED	turtle excluder device
TEWG	Turtle Expert Working Group
TGL	Texas Gulflink
TL	total length
UFC	Unified Facilities Criteria
U.S.	United States
USACE	United States Army Corps of Engineers
USCG	United States Coast Guard
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USVI	United States Virgin Islands
VLCC	Very large crude carriers
VOC	volatile organic compounds
VPM	Vapor Processing Module
WFGB	West Flower Garden Bank
WCD	worst case discharge
WCS	Western Canadian Select heavy grade crude oil
WPA	Western Planning Area
WTI	West Texas Intermediate light crude oil
ZOI	zone of influence

UNITS OF MEASUREMENT

ac	acre(s)
bbl	barrel (of oil)
°C	degrees Celsius
cm	centimeter(s)
°F	degrees Fahrenheit
ft	foot/feet
ft^2	square foot/feet
g/m^2	grams/square meter
in	inch(es)
km	kilometer(s)
lin ft	linear foot/feet
m	meter(s)
mi	mile(s)
mi^2	square mile(s)

INTRODUCTION

Section 7(a)(2) of the ESA of 1973, as amended (16 U.S.C. § 1531 et seq.), requires that each federal agency ensure that any action authorized, funded, or carried out by 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 National Oceanic and Atmospheric Administration (NOAA) National Marine Fisheries Service (NMFS) and the United States Fish and Wildlife Service (USFWS) share responsibilities for administering the ESA.

Consultation is required when a federal action agency determines that a proposed action “may affect” listed species or designated critical habitat. Informal consultation is concluded after NMFS determines that the action is not likely to adversely affect listed species or critical habitat. Formal consultation is concluded after NMFS issues a Biological Opinion (hereafter, referred to as the Opinion) that identifies whether a proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify critical habitat, in which case reasonable and prudent alternatives to the action as proposed must be identified to avoid these outcomes. The Opinion states the amount or extent of incidental take of listed species that may occur and develops nondiscretionary measures to reduce the effect of take. The Opinion may also recommend discretionary conservation measures to further the recovery of the species. No incidental destruction or adverse modification of critical habitat may be authorized. The issuance of an Opinion detailing NMFS’s findings concludes ESA Section 7 consultation.

This document represents NMFS’s Opinion based on our review of impacts associated with the proposed actions to issue permits for the construction, operation and decommissioning of two deep water port projects within Brazoria County, Texas. This Opinion analyzes the projects’ effects on threatened and endangered species and designated critical habitat, in accordance with Section 7 of the ESA. We based it on project information provided by the United States Coast Guard (USCG), Maritime Administration (MARAD), and other sources of information, including the published literature cited herein.

On July 5, 2022, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR part 402 in 2019 (“2019 Regulations,” see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court’s July 5 order. As a result, the 2019 regulations are once again in effect, and we are applying the 2019 regulations here. For purposes of this consultation, we considered whether the substantive analysis and conclusions articulated in the biological opinion and incidental take statement would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

1 CONSULTATION HISTORY

Sea Port Oil Terminal (SPOT) Deepwater Port Project

We received a letter from the USCG requesting consultation on January 13, 2020. On May 26, 2020, NMFS sent comments to the USCG and MARAD on the SPOT project Draft Environmental Impact Statement. NMFS requested additional information on February 26, 2020, February 17, 2021, and March 2, 2021. NMFS received a final response on April 8, 2021, and initiated consultation that day. During the internal review of the Biological Opinion, NMFS requested additional information on November 21, 2021, April 15, 2022, and April 22, 2022. NMFS received a final response on April 25, 2022.

Texas GulfLink (TGL) Deepwater Port Project

On June 27, 2019, the USCG sent an email requesting NMFS's assistance with the review of the TGL deepwater port application to determine the completeness and sufficiency of the TGL application. NMFS agreed and participated in several interagency calls and email exchanges with USCG, TGL, and third party consultants. On May 11, 2020, USCG sent an email requesting that NMFS provide pre-consultation technical assistance for the TGL Deepwater Port Project, in preparation for the upcoming ESA Section 7 consultation. NMFS agreed and proceeded to participate in additional interagency calls, email exchanges and document reviews for the project. On December 1, 2020, USCG and MARAD sent NMFS a letter (via email) requesting concurrence with its determination that the TGL project was not likely to adversely affect ESA-listed species under NMFS purview. Following another series of calls and email exchanges, on March 3, 2021, NMFS requested additional information on pile driving and noise abatement measures for the project. On March 18, 2021, NMFS received a partial response to the information request, and on March 20, 2021, received the remainder of the information necessary to complete our effects analysis, and consultation was initiated that day. During the consultation process NMFS decided to batch the TGL and SPOT Deepwater Port project consultations into a single consultation document. This decision was based on the similarities between the projects, geographic locations, timing of initiation of consultation, and effects to ESA-listed species and designated critical habitat under NMFS's purview.

2 DESCRIPTION OF THE PROPOSED ACTIONS AND ACTION AREA

2.1 Proposed Actions

2.1.1 Sea Port Oil Terminal (SPOT) Deepwater Port Project

SPOT Terminal Services LLC (hereafter referred to as SPOT), a wholly owned subsidiary of Enterprise Products Operating LLC (EPO), is proposing the SPOT Deepwater Port (DWP) Project (hereafter referred to as the SPOT DWP). On January 31, 2019, SPOT submitted an application to the United States Coast Guard (USCG) and Maritime Administration (MARAD) seeking a Federal license under the Deepwater Port Act of 1974 (DWPA), as amended, to own, construct, operate, and eventually decommission a DWP for the transportation of crude oil for export to the global market. The SPOT DWP would be capable of exporting various types of crude oil, including ultralight crude oil, such as processed condensate; light crude oil, such as the West Texas Intermediate (WTI); and heavy grade crude oil, such as Western Canadian Select (WCS).

The proposed SPOT DWP will be located in Federal waters of the Gulf of Mexico (GoM), in the Galveston Area Outer Continental Shelf (OCS) lease blocks 463 and A-59, approximately 27.2

to 30.8 nautical mi (nmi) off the coast of Brazoria County, Texas, in water depths of up to 115 feet (ft). Figure 1shows a diagram of the specific SPOT DWP and its components. The offshore components of the Project will consist of:

• Two co-located, 36-inch (in)-diameter crude oil offshore pipelines for crude oil delivery to very large crude carriers (VLCCs) or other crude oil carriers from the terminal;
• One fixed offshore platform (Figure 1) with eight piles, four decks, and three vapor combustion units;
• Two single point mooring (SPM) buoys to moor the VLCCs or other crude oil carriers for loading;
• Four pipeline end manifolds (PLEMs) (two per SPM buoy) to provide the interconnection between the crude oil pipelines and the vapor recovery pipelines, and the SPM buoys;
• Four 30-in-diameter pipelines (two per PLEM) to deliver crude oil from the platform to one of the two PLEMs at each SPM buoy;
• Four 16-in-diameter vapor recovery pipelines (two per PLEM) to connect to one of the two PLEMs at each SPM buoy to the three vapor combustion units on the platform;
• Three service vessel moorings, located in the southwest corner of Galveston Area lease block 463 and;
• An anchorage area in Galveston Area lease block A-59, which will not contain any infrastructure.

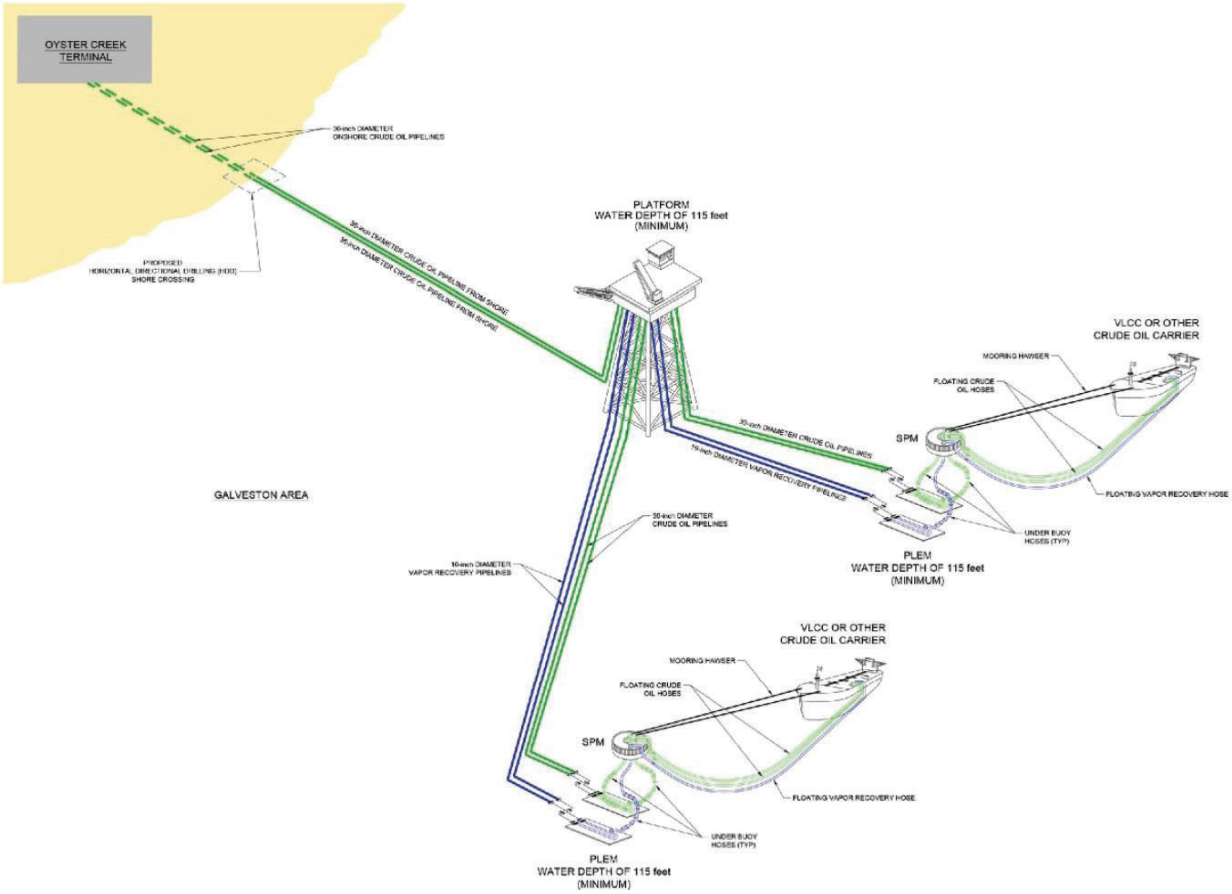
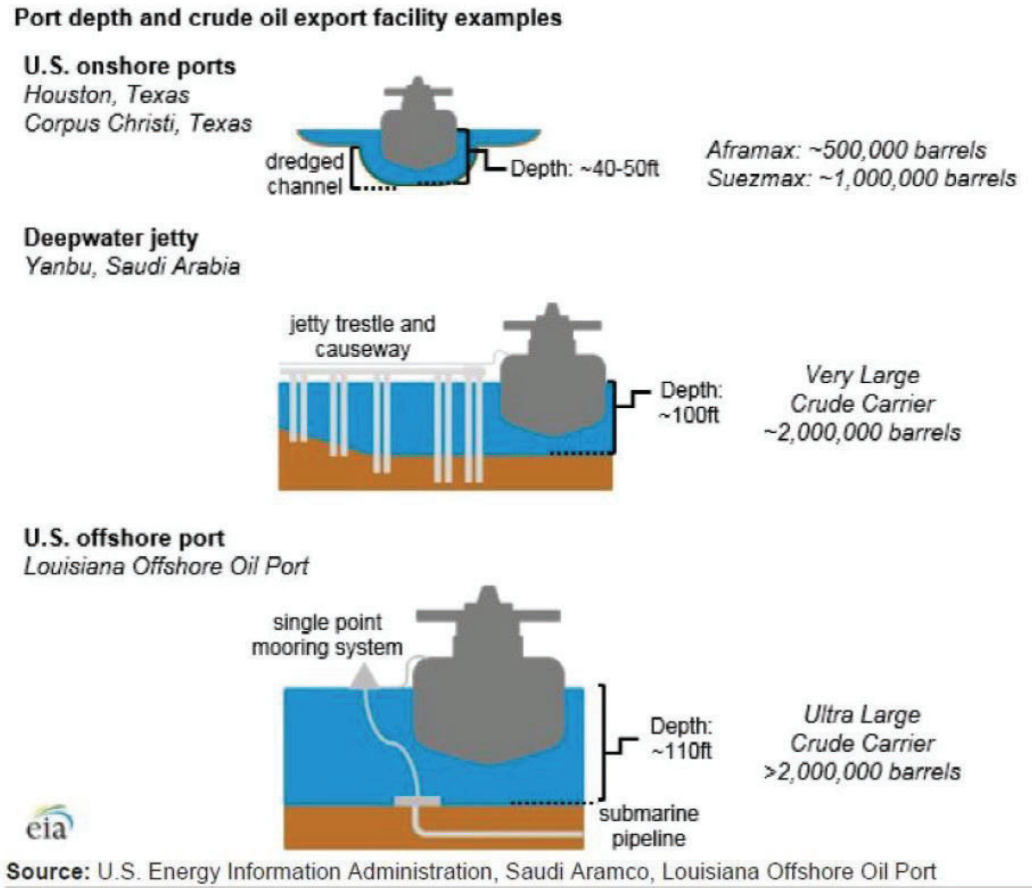


Figure 1. SPOT DWP and components diagram (Figure 2.2-4 in the Sea Port Oil Terminal Deepwater Port Project Final Environmental Impact Statement [SPOT FEIS], July 2022)

Onshore and offshore pipelines will deliver mostly domestically produced crude oil to the SPOT DWP for loading and export via VLCCs or other crude oil carriers. EPO proposes to use its affiliates' existing assets and access to crude oil supplies from multiple sources along the northern Texas Gulf Coast. Crude oil, ranging from ultralight to heavy grade, will be transported by two interconnected, co-located offshore pipelines. The offshore pipelines will deliver crude oil to the SPOT DWP. Various grades of crude oil, at flow rates of up to 85,000 barrels per hour (bph), will be loaded to VLCCs or other crude oil carriers moored at the two SPM buoys. The maximum frequency of loading VLCCs or other crude oil carriers will be 2 million barrels per day (bpd), 365 days per year. VLCCs have a maximum capacity of 330,693 U.S. tons and a maximum draft of approximately 71 ft. Other crude oil carriers that could call on the SPOT DWP include the Suezmax and the Aframax. The Suezmax has a maximum capacity of 242,508 U.S. tons and a draft of 66 ft. The Aframax has a maximum capacity of 132,277 U.S. tons and a draft of 49 ft. Figure 2. Relative Ship and Channel Sizes illustrates relative ship sizes and channel depths.



Source: EIA 2018b

Figure 2. Relative Ship and Channel Sizes (Figure 1.1-4 in the SPOT FEIS, July 2022)

Subsea Pipeline Description and Installation

SPOT will construct two co-located 36-in-diameter pipelines to deliver crude oil from the Oyster Creek to Shore Pipelines to the SPOT DWP. Each pipeline will be approximately 40.8 nmi long. The pipelines would be oriented perpendicular to the shoreline towards the southeast, then turn to run northeast and then southeast around the Freeport Harbor anchorage area. After skirting the anchorage area, the pipelines would run southeast across the Texas state water and federal water boundary and the Aransas Pass to Calcasieu Pass Shipping Fairway 3, and turn 90 degrees southwest to perpendicularly cross the Freeport Shipping Fairway. The subsea pipelines would turn south through five lease blocks to the proposed SPOT DWP in Galveston Area lease block 463. Figure 3 shows the SPOT DWP pipeline route and the lease blocks the pipelines would cross.

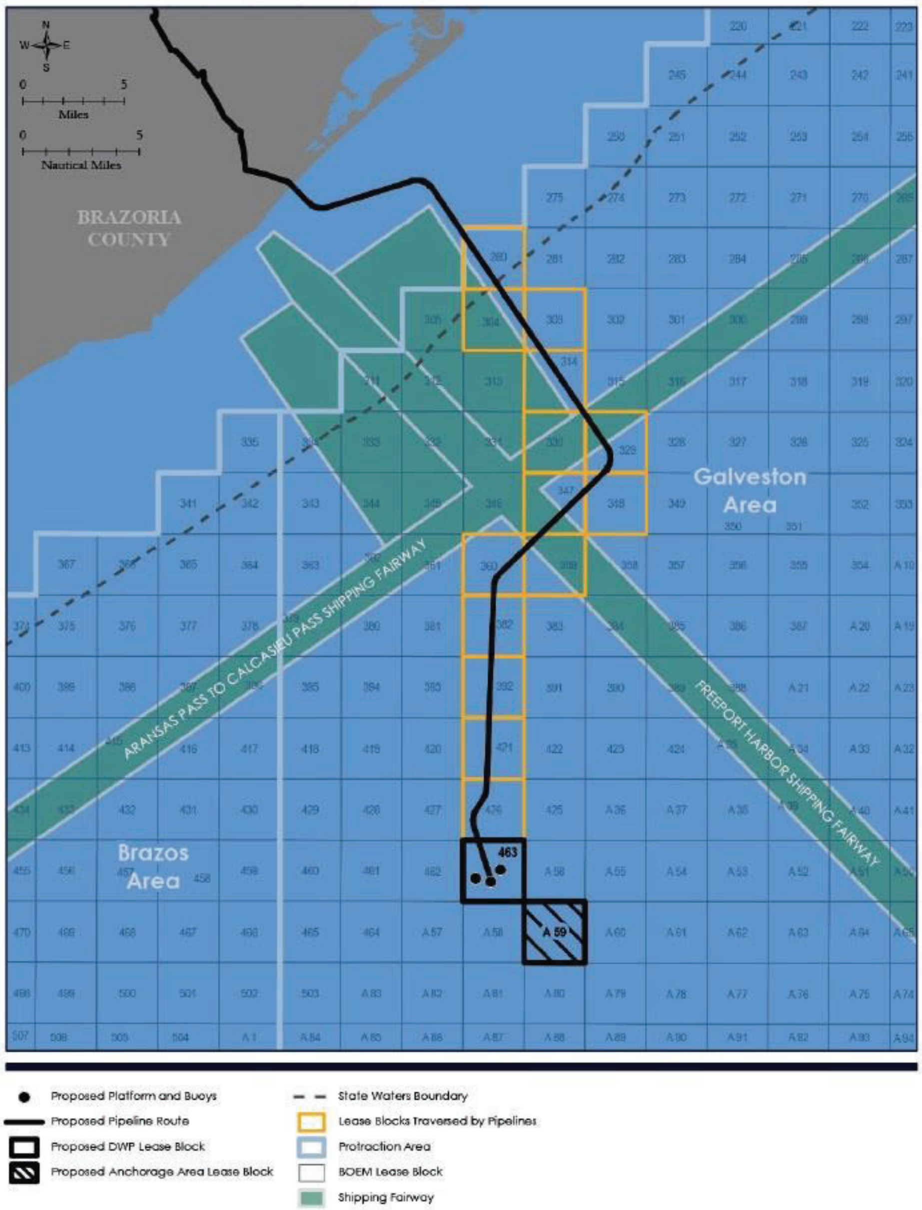


Figure 3. SPOT DWP Lease blocks and pipeline route (Figure 2.2-3 in the SPOT FEIS, July 2022)

The subsea crude oil pipelines will conform to the definitions of American Society of Mechanical Engineers (ASME) 600#, and will be concrete coated to approximately 5 in for on-bottom stability. The pipelines will also be coated with 15 to 16 mils (1 mil equals 0.001 in) fusion-bonded epoxy for corrosion protection. The pipelines will be installed a minimum of 3 ft below the ocean floor; however, in areas where the pipelines would cross a shipping fairway, the pipelines would be installed a minimum of 10 ft below the underwater natural bottom elevation (49 CFR § 192.327, 49 CFR § 192.612). Subsea pipelines will be buried using a jet sled. The SPOT DWP project also proposes utilizing existing pipelines and cables that are active. Existing pipelines and cables will be lowered using high-pressure water jets and a compressed air-lift operation to remove any cover above the existing pipelines or cables. The same water jetting air-lift operation will be used to lower the existing pipelines or cables to a depth that will allow a minimum of 18-in separation between the new 36-in diameter crude oil export pipeline and the existing pipeline or cable and a 3 ft cover over the top of the new pipeline below the natural seabed. Where the subsea pipelines would cross existing pipelines, concrete mattresses will be added for separation and protection, and bracelet anodes for cathodic protection would be installed.

The subsea pipelines will have devices (referred to as pigs) that can perform various maintenance operations without the need to stop the flow of oil, referred to as pigging capability. The stations that launch and receive the pigs will be located within the Oyster Creek Terminal, allowing continuous pigging (i.e., cleaning) from the Oyster Creek Terminal to the SPOT DWP platform and back. The subsea crude oil pipelines will be installed from the shore crossing to just over 1.0 mi (5,500 ft) offshore using the Horizontal Direction Drill (HDD) method. HDD is a trenchless method in which a pit is dug on either side of an obstacle. In the case of HDD, the horizontal hole is drilled using a pilot hole, and then a series of sequentially larger diameter reamers to increase the size of the boring to the desired size. The hole is kept open by drilling mud composed of water and bentonite. Once the hole is complete, the carrier pipe is connected to the pipeline and pulled through the hole. To accomplish this, the pipeline segments must be welded together on the surface and laid in line with the borehole. Therefore, HDD installation often requires a temporary, linear work area for constructing the pipeline before it is pulled through the boring.

From the HDD exit site, the pipelines will be installed using a lay barge and trenched with a sled. SPOT selected the HDD method for installing the subsea pipelines from the shore crossing running offshore to avoid impacts on the beach area at the shoreline, as well as nearshore waters and habitats that experience high levels of commercial and recreational use. The co-located subsea pipelines will be bi-directional, allowing different crude oil grades to be transported to and from the platform. SPOT will change crude oil grades in the pipelines by displacing one product from one pipeline through the second pipeline back to the Oyster Creek Terminal. This will allow the platform to load two different crude oil grades to two VLCCs or other crude oil carriers concurrently, or load the vessels with the same product back to back.

The HDD method will involve a survey crew marking the location of the HDD exit hole (offshore) and placing tracking coils. A spud barge, or anchored barge, with bucket dredge will excavate the exit pipe to provide a vertical wall for the pilot hole to punch through. When pulled into position, an adequate length of pipe on the seabed will allow the pipeline installation barge to recover into the lay system.

Upon completion of reaming, the start-up head of the pipe section will be attached to the drill string using a wire rope sling and swivel. The support barge will assist and align the pull section to reduce axial tension loads. During the pull-back process, the pipeline installation barge will follow the pipe string with an abandonment and recovery cable attached to the laydown assembly to help keep the tail of the pipeline from drifting from the proposed alignment due to currents and tides. There will be two offshore exit area workspaces and each will be about 400 feet long x 50 feet wide x 20 feet deep. The distance between the co-located pipelines onshore at the HDD entrance is 30 ft. The distance between the co-located pipelines at the HDD exit point in the GoM is 164 ft. Starting at the HDD exit point, each subsea pipeline will be strung individually, roughly parallel to each other. Distance between pipelines at offshore platform is 164 ft. The subsea pipelines will be buried consecutively using a jet sled.

Throughout the drilling process, a slurry of non-toxic, bentonite clay and water will be pressurized and pumped through the drilling head to lubricate the drill bit, remove drill cuttings, and hold the hole open. Special additives may also be required, typically during the pilot hole phase, but would constitute a small fraction of the drilling fluid, which is generally considered to have low toxicity. The slurry, referred to as drilling mud or drilling fluid, has the potential to be inadvertently released to the surface if fractures or fissures occur, or during the drilling of the pilot hole when the pressurized drilling mud is seeking the path of least resistance. The path of least resistance is typically back along the drilled pilot hole. However, if the drill path becomes temporarily blocked or large fractures or fissures that lead to the surface are crossed, then an inadvertent release could occur. The drilling construction contractor will monitor the pipeline route and the circulation of drilling mud throughout the HDD operation for indications of an inadvertent drilling mud release and will immediately implement corrective actions if a release is observed or suspected, such as establishing containment structures where necessary and working with regulatory agencies in accordance with applicable regulations and permit conditions to determine the necessary course of action. After the HDD punches out, drilling fluid will settle on the bottom of the construction pit and will be recovered with a diver and suction hose. The suction hose will transport the drilling fluids to a tank located on a vessel. Once the drilling fluids within the tank have settled out, the material will be properly disposed of at a locally approved land fill or disposal facility.

Pipeline Construction

The HDD technique will be used to install the two co-located pipelines between the onshore and offshore segments. The HDD entry hole will be located onshore and the exit pit or trench will be excavated offshore near the 25- to 28-ft water depth contour or about 5,500 ft from the shoreline. Spoil materials will be sidecast within the temporary workspace on either side and on the shore side of the trench. The HDD exit hole will be allowed to naturally backfill due to movement from currents, tides, and wave action.

A pipeline installation barge will install a start-up anchor approximately 200 ft from the planned HDD exit hole at about the 30-ft water depth contour and begin assembling the HDD pipe string. The barge will move forward once each pipe joint is welded together on the installation barge. The completed 7,500-ft-long pipe string will be laid on the seafloor and an anchor will be installed on the deep end to hold the pipe in place. The process would be repeated for the second pipeline.

The HDD drilling rig will operate from the shore side and a reaming support barge will operate offshore. The pipe installation barge and a support barge will assist in pullback operations once reaming is complete. After pullback of the two 36-in-diameter pipe segments is completed, the ends would be secured with anchors. Each pipeline segment will be filled with seawater and hydrostatically tested.

The remaining sections of the two 36-in-diameter offshore pipelines will be installed using a conventional, anchored pipeline installation barge (pipelay barge). This method uses cargo barges and tugs to transport pipe joints to the installation barge where pipe joints will be welded, inspected, and field joint coatings will be applied. Work will begin near the HDD exit hole and will use anchor handling tugs that will position and hold the pipeline installation barge along the right-of-way using two stem anchors, a minimum of two bow anchors, and four breast anchors. As pipe segments are completed, the pipelay barge will move forward until the entire pipeline is laid on the seafloor. The same process will be repeated for the second pipeline.

The pipelay barge will install the four 30-in-diameter loading pipelines between the platform and PLEM target box locations at each SPM buoy. A deadman anchor would be set in line with the pipeline route and the pipelay barge would assemble and lay the pipe moving away from the startup anchor as described above.

Once the design length of the pipeline is welded, a flanged laydown head will be installed and the pipe will be lowered to the designated location on the seafloor. The four 16-in vapor recovery pipelines will be installed in the same manner between the platform and PLEM locations at each SPM buoy.

Upon completion of pipeline installation on the seafloor, a trenching vessel using a jet sled would be positioned at the HDD point and use high-pressure water jets to break up the consolidated bottom materials alongside and underneath the pipeline. High-pressure compressed air will remove the slurry beneath the pipe as the barge moves ahead. The substrate hardness would determine the rate of travel. The same process will be followed for the two 36-in, four 30-in, and four 16-in diameter pipelines. The pipelines will be buried to a minimum depth of 3 ft below the seafloor except at shipping fairway crossings, which will require a burial depth of 10 ft below the seafloor. Multiple passes may be required to achieve this depth.

Where the pipeline will cross existing pipelines and cables, high-pressure water jets and compressed air-lift operation will be used to remove any cover above the existing pipelines and cables. The same jetting and air-lift operation will then be used to lower the existing pipelines or cables to a depth that will allow a minimum of18-separation between the existing pipelines or cables and the new pipelines and a 3-foot cover over the top of the new pipeline below the natural seabed. Concrete mats would be placed on top of the existing pipeline or on either side of existing cables to maintain the 18-in separation. Concrete mats or sandbags would be placed over the new pipeline in areas where 3 ft of cover could not be achieved due to existing pipeline elevations.

SPOT will use the Transportation Security Administration Pipeline Security Guidelines to decrease the chances of terrorism and sabotage. SPOT will include in the project operations manual safety plans or risk mitigation measures to respond to natural forces and weather-related incidents that could affect the pipeline and offshore platform.

Offshore Platform Components and Pipeline End Manifolds (PLEMs)
The SPOT DWP will be a manned jacket-designed platform. The jacket is the structure that supports the decks and all equipment, and will be supported with eight 72-in diameter steel piles. The jacket structure would be anchored by driving piles within the jacket structure legs. The jacket itself will be installed first and will serve as the confined system through which

compressed air will be introduced between the piles and the jacket legs to minimize underwater noise impacts during construction (Figure 4, Jacket system).

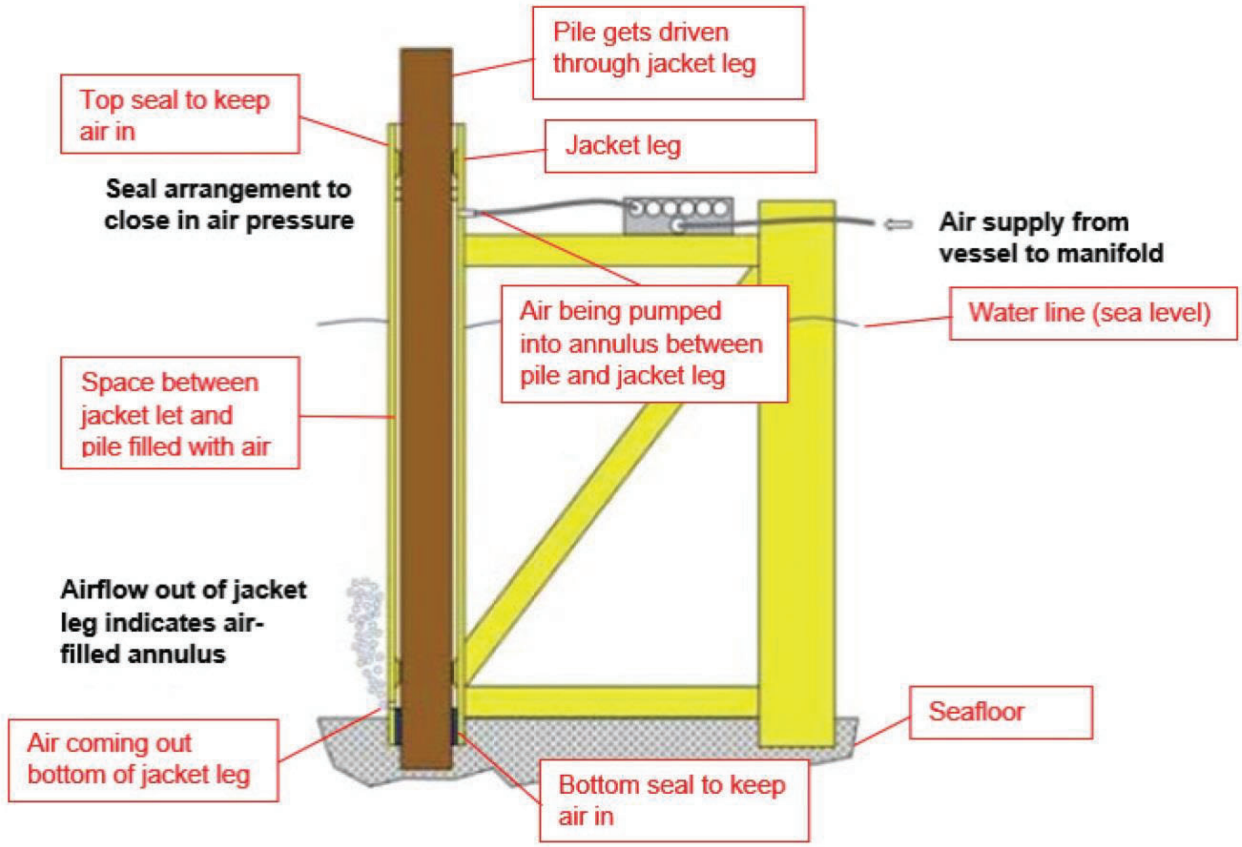


Figure 4. SPOT DWP Pile Jacket System (Figure 2.2-6 in the SPOT FEIS, July 2022)

Table 1 outlines the piles that will be used for the SPOT DWP. The jacket will be fabricated off site and brought to the SPOT DWP via cargo barge. Jacket and pile installation will occur prior to deck installation. The jacket will be lifted and set in position, then verified by an on-site surveyor. The 72-in-diameter platform piles will be driven through the jacket legs and into the sea floor, to a depth of 380 ft below sea bottom elevation using a pile driver hammer operating from a derrick barge. Platform piles will require 1,278 strikes per hour (hr) while the impact hammer is in operation (2 hrs out of every 6 hrs). The pile driving operations will occur 24 hrs per day, resulting in approximately 10,224 strikes per (24-hour) day. Installation of the eight 72-in diameter piles would take about 10 days. Each pile will consist of four sections, which will be welded together (end-to-end) as each section is driven, resulting in a total length of 514 ft per pile. Once the jacket is fully secured in place by the platform piles, each deck will be set on top of the jacket legs, and then welded in place.

Table 1. Pile-Driving Details for the 2 Types of Structures

Parameter	Piles Supporting the Offshore Platform	Piles Supporting the PLEMs
Pile type	steel pipe	steel pipe
Pile number	8	16
Pile diameter	72 inches	30 inches
Pile-driving method	impact	impact
Number of strikes per pile	10,224	408
Driving time per pile	24 hrs	8 hrs
Number of piles per day	Up to 1	3
Total strikes per day	10,224	1,224
Number of driving days	10	6
Attenuation device	confined bubble curtain	unconfined bubble curtain
Attenuation rate	10 dB	5 dB

Tie-in spools will be fabricated at onshore facilities and transported to the installation location by a supply vessel. A dive support vessel will lower the tie-in spools to the seafloor and the flanged ends will be connected between the pipelines, the risers, and the PLEMs. Flanged connections with swivel and misalignment ball flanges will be used for installation, as required, to facilitate the connection of the offshore pipelines to the fixed orientation of the jacket risers.

The PLEMs will be transported on a material transport barge and will be lowered to the ocean floor with support from a dynamic positioned diving support vessel. The 4 PLEMs will be secured in place with four 30-in diameter steel piles each. Pile driving will occur 24 hrs per day with each pile requiring 408 strikes over 8 hrs to complete the driving process. It will take approximately 10 days to install all of the 16 piles.

The Applicant proposes to use an unconfined, single-ring bubble curtain around the PLEM piles (Figure 5 and Figure 6).

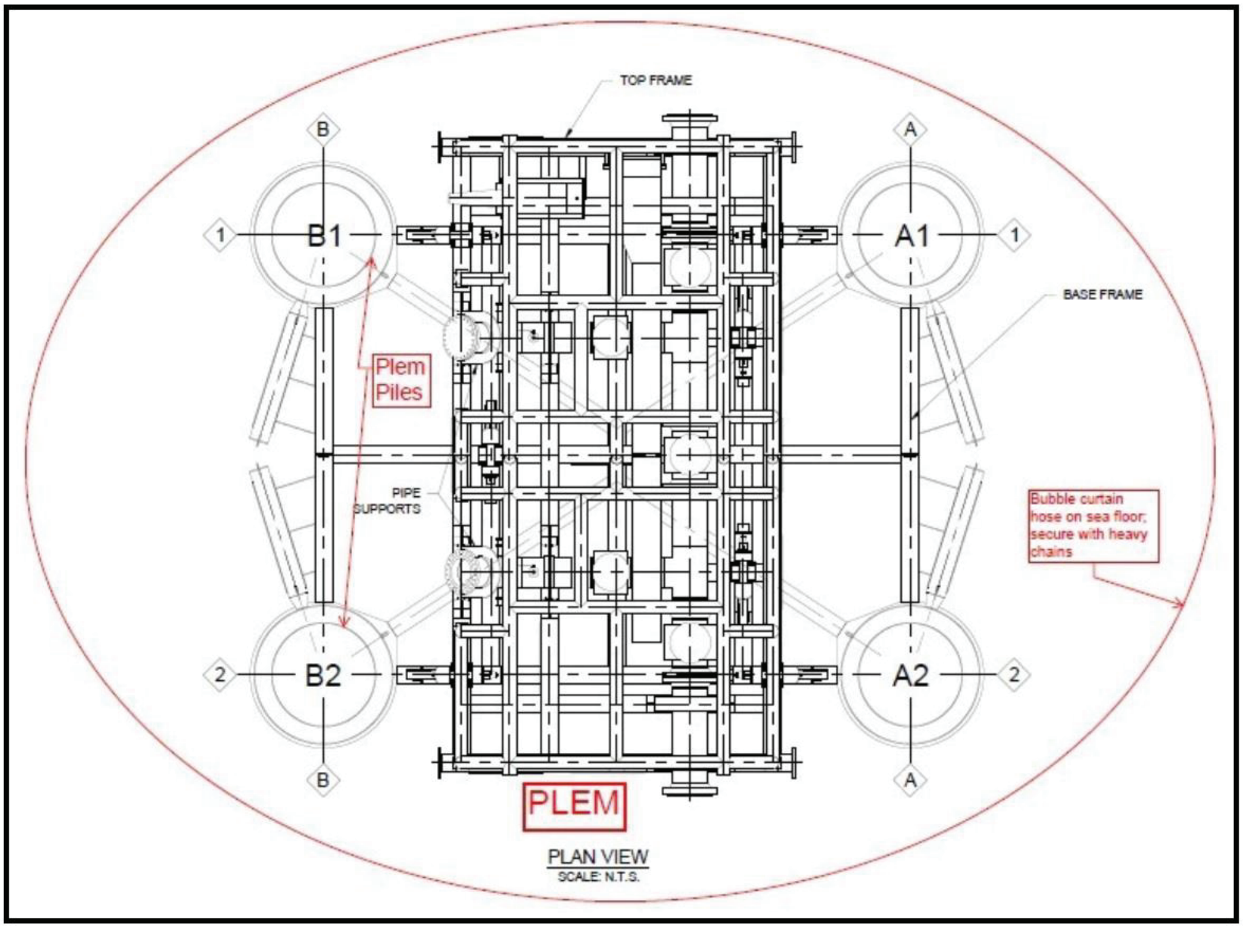


Figure 5. An unconfined, single-ring bubble curtain system for the 30-in PLEM piles (Figure 341-2 in the SPOT Deepwater Port License Application Data Gap Response #14 – Part A, April 2021)

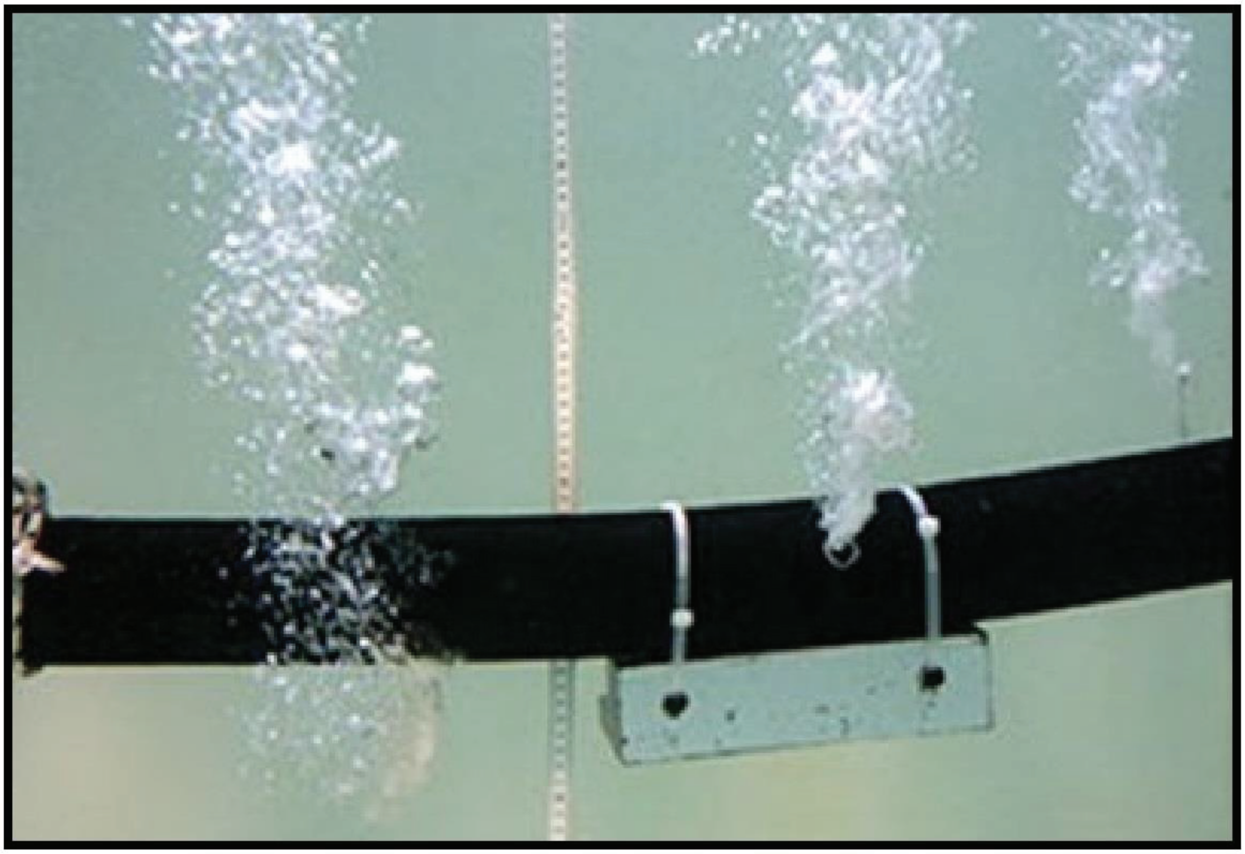


Figure 6. Image of an operating, weighted, unconfined, single-ring bubble curtain system for the 30-in PLEM piles (Figure 341-3 in the SPOT Deepwater Port License Application Data Gap Response #14 – Part A, April 2021)

The SPM system will use fluke anchors and anchor chains to secure the buoy in position. The six anchors would be equally spaced on a 1,043-foot radius circle with 1,080 ft of anchor chain between the anchor and the chain stopper on the buoy. An installation vessel will first install each fluke anchor and lay out the chain. A large anchor handling tug will set the anchors by pulling the anchor in the direction of the buoy's proposed location, then laying the chain on the seafloor. After the pile anchors and anchor chains are laid and inspected, the SPM buoy will be towed into the designated location and the anchor chains will be installed in accordance with the buoy designer's recommended installation sequence and procedures. After inspection, the underbuoy hoses would be installed following the Oil Companies International Marine Forum guidelines. Once the SPM buoy installation is complete, including the installation of the underbuoy hoses to the PLEM, the SPM buoy system will be fully inspected.

Relevant Best Practices (BPs) for pile driving procedures include:
(1) Acoustic Disturbance Mitigation

a. SPOT will implement a "soft start" procedure to pile driving, which involves ramping up the intensity of the hammer strikes prior to operating at full capacity.

b. A bubble curtain system will be applied to all pile driving activity.

c. The Applicant will implement a “shut down” of pile driving activity if a marine mammal is observed approaching or within the ensonified zone for level A Harassment (the area of acoustic effects that can produce injury to marine mammals).

(2) Visual Monitoring

a. During pile driving activities a NMFS-approved Protected Species Observer(s) (PSO) will be stationed onboard the pile driving vessel and conduct monitoring for protected species within a pre-determined zone of influence (ZOI).
b. Prior to the start of in-water activities, the PSO will monitor the ZOI for 30 minutes to ensure that the area is clear of all ESA-listed species and marine mammals. The activity would only commence once the PSO has declared that the ZOI is clear.
c. The PSO would monitor the ZOI for the entirety of the in-water activity and record any sightings of protected species. Monitoring will be conducted during daylight and nighttime hours to account for 24-hr pile driving activity.
d. Observations will be conducted using high-quality binoculars during daylight hours, and with thermal imaging cameras or night vision binoculars during periods of low or no light situations (e.g., nighttime pile driving).
e. All marine mammal and ESA-listed species sightings would be fully documented.
f. Following the in-water activity, the PSO would conduct post-monitoring of the ZOI for 30 additional minutes.

PLEMs and Interconnection Pipelines

The offshore platform will connect to the SPM buoys by a series of pipelines running through four PLEMs. Underneath each SPM buoy, two PLEMs will be installed on the ocean floor. One PLEM at each SPM buoy will be connected to the platform via two 30-in crude oil loading pipelines; approximately 0.66 nmi in length. In addition, one PLEM at each SPM will connect to two 16-in vapor recovery pipelines of the same length as the crude oil pipelines running between the PLEMs and the platform.

The crude oil loading PLEMs will allow for the transfer of crude oil from the platform, through the crude oil loading pipeline, and into the crude oil floating hose to the VLCC or other crude oil carrier for loading. The vapor recovery PLEMs will facilitate the transfer of vapors created during the loading process from the VLCCs or other crude oil carriers through the vapor recovery floating hoses, into the vapor recovery pipelines, and back to the platform. SPOT will use 30-in steel foundation piles to install the PLEMs on the ocean floor. Each PLEM will have pigging capabilities and valves for maintenance purposes.

Single Point Mooring Buoys and Interconnection Hoses

The SPOT DWP will include two SPM buoys to moor VLCCs or other crude oil carriers concurrently. The SPMs are floating buoys, which will be anchored in the same lease block as the platform 0.66 nmi away, and will connect to the PLEMs by floating crude oil and vapor recovery hoses. Both SPMs will be anchored in approximately 115 ft of water, using six fluke anchors per SPM connected by anchor chain. Fluke anchors are anchors with triangular projections that help the anchor become buried in the ocean floor and keep the buoys in place.

The fluke anchors will be installed in a circular manner around the SPM buoys, each separated by 60 degrees, and connected by the anchor chain. The fluke anchors will be designed for calculated loading conditions and site-specific geotechnical conditions. This configuration will allow the SPM buoys to move as needed based on wind, waves, current, and VLCC or other crude oil carrier conditions, within defined limits. The anchor system for each SPM buoy will disturb 0.182 acre (ac) of seafloor, totaling 0.364 ac of seafloor for both SPM buoys. Figure 7shows an example of an SPM with moored vessel.

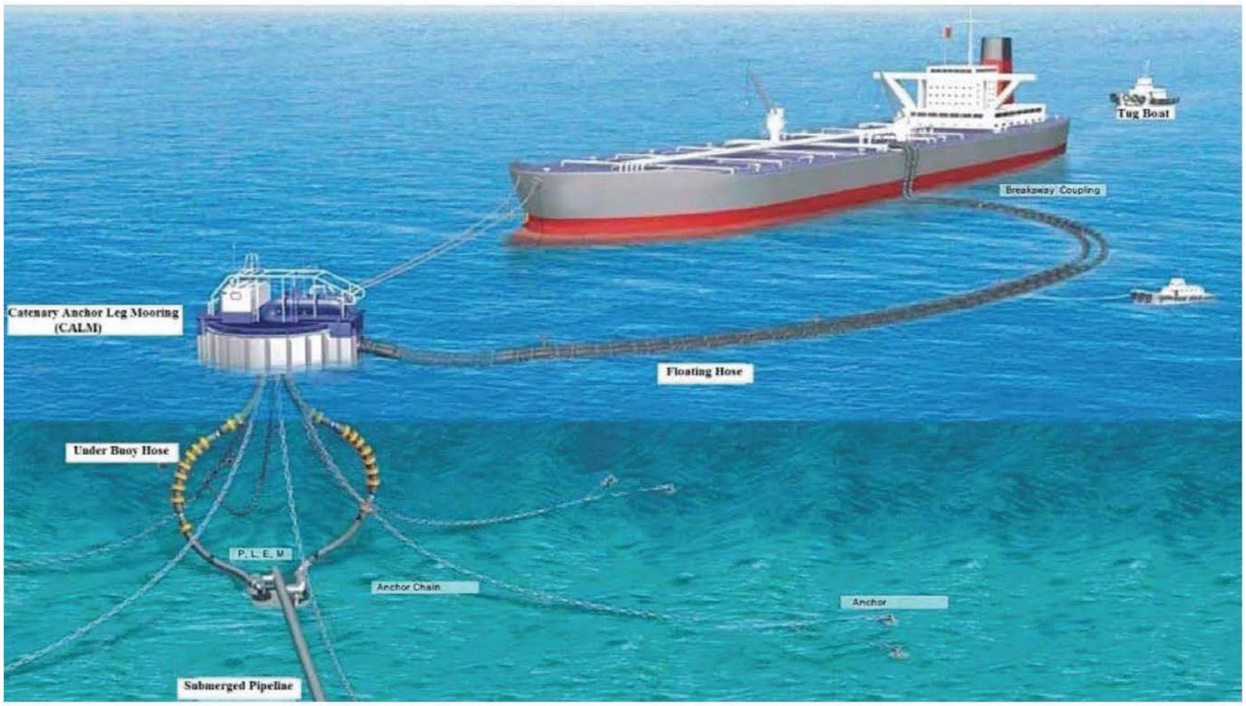


Figure 7. SPM Buoy Mooring Schematic (Figure 2.2-7 in the SPOT FEIS, July 2022)

Each SPM buoy will connect to one PLEM via two 24-in underbuoy crude oil hoses and another PLEM via one 24-in underbuoy vapor recovery hose. Once crude oil reaches the SPM buoy, it will be loaded on the VLCC or other crude oil carrier via two 24-in crude oil floating hoses. Each floating crude oil hose will include approximately 200 ft of 16-in tail hose that will be lifted and hang over the edge of the railing of the vessel being loaded.

Each SPM buoy will also include one 24-in vapor recovery hose of approximately 1,000 ft in length connected to the moored VLCC or other crude oil carrier. Similar to the floating crude oil hoses, each vapor recovery hose will include approximately 200 ft of 16-in tail hose. This floating hose will connect with the underbuoy vapor recovery hose leading back to the PLEM. During normal sea states, the floating hoses will typically float parallel to each other and avoid entanglement. During high or turbulent sea states, the floating hoses could become entangled with each other or with marine debris. Prior to each loading of crude oil on a VLCC or other crude oil carrier, SPOT will ensure the hoses are untangled. Because a maximum of 365 total loadings are anticipated per year, frequent (i.e., daily) handling or untangling of hoses is expected to minimize potential hose damage from long-term entanglement.

Hydrostatic Testing

Approximately 14 million gallons of seawater will be used for hydrostatic testing of offshore pipelines. Corrosion inhibitors will be added to the water during testing and, therefore, released into the GoM upon completion. SPOT anticipates using a corrosion inhibitor with propylene glycol and polyoxyalkylenes. No information about polyoxyalkylene toxicity is available, but propylene glycol has been shown to be relatively non-toxic in marine and freshwater environments and is highly water soluble. Oxygen is required for organisms to metabolize propylene glycol, which can lead to low dissolved oxygen concentrations at release sites (Canadian Council of Ministers of the Environment 2006). SPOT anticipates withdrawing seawater at a rate between 5,800 and 14,600 gallons per minute (gpm). Seawater will be filtered through one or more sieves with a final mesh screen no coarser than 5/16-in, capable of removing 99 percent of all particles greater than or equal to 92 microns in diameter.

After pressure testing is complete, the pipeline will be dewatered, cleaned, and dried, using air to run a series of pipeline pigs through the system. Upon completion, the hydrostatic test, seawater will be discharged at a rate of 4,000 gpm, and will take approximately 60 hrs. The discharge of seawater will occur via the platform deck drain, which flows back to the GoM.

Construction Vessels

SPOT estimates that a total of 33 vessels will be needed during construction, but there would be less than that used at any one time. Construction vessel speeds will vary. Barges and tugs will generally be intermittently stationary or moving at speeds of 14 knots or less during the project component installations. Smaller support vessels of 16 to 49 ft could reach speeds of up to 35 knots, especially when transporting crews or supplies to and from the SPOT DWP project area.

To reduce the risk of a vessel strike during the SPOT DWP construction, all construction vessels will comply with NOAA's NMFS Southeast Region's Protected Species Construction Conditions (NMFS 2021). These conditions require, among other things, that operations of moving equipment shall cease if a protected species is observed within 150 feet of operations and that activities shall not resume until the protected species has departed the project area of its own volition (e.g., species was observed departing or 20 minutes have passed since the animal was last seen in the area). Construction vessels will also comply with NOAA Fisheries' Vessel Strike Avoidance Measures (NMFS 2021). These measures require, among other things, that vessel operators and crew maintain a vigilant watch for protected species (i.e., marine mammals and ESA-listed species) to avoid striking sighted protected species. In addition, these measures also require all vessels operate at "idle/no wake" speeds at all times while in the project construction area, while in water depths where the draft of the vessel provides less than four feet of clearance from the bottom, and in all depths after a protected species has been observed in and has recently departed the area.

Operation Vessels

SPOT anticipates a maximum of 365 vessel calls per year by VLCCs or other crude oil carriers. SPOT estimates this will roughly double the vessel traffic in Galveston Area lease block 463 (based on vessel traffic reported during 2016 and 2017 for this lease block).

A number of service vessels would be required to assist with operation of the SPOT DWP, such as tugboats, supply vessels, and crew boats. SPOT will would provide three mooring points for these vessels in the southwest corner of Galveston Area lease block 463. These service mooring points will be anchored to the ocean floor using two concrete blocks of 20,000 pounds each, joined together with a bridle. An anchor chain will attach to the bridle and connect to the mooring buoy. The total disturbance of all three service vessel moorings will be 0.0016 ac. SPOT proposes to designate a self-enforced precautionary area of a specified distance around each service vessel mooring point to allow the vessels to turn or move with the ocean currents as necessary. This will meet guidelines established by the World Association for Waterborne Transport Infrastructure, also known as PIANC. Each mooring point will have a secondary precautionary area of a prescribed distance to minimize risk of collision, which will meet Unified Facilities Criteria established by the U.S. Department of Defense (UFC 4-150- 06) and will be administered by SPOT. A discussion regarding the types of service vessels that would be used during operation of the SPOT DWP can be found in Section 2.2.4, Service Vessels and Helicopters, of the SPOT Supplemental Draft Environmental Impact Statement (SPOT S-DEIS).

All SPOT DWP-related activities will comply with Federal regulations to control the discharge of operational wastes, such as bilge and ballast waters, trash and debris, and sanitary and domestic waste generated from vessels and platforms associated with the proposed SPOT DWP. VLCCs and other crude oil carriers may discharge water associated with ballast and cooling water into the GoM while at the SPOT DWP, but domestic or sanitary water onboard the VLCCs or other crude oil carriers will be held in storage tanks. Vessels that release oil or oily ballast water at the SPOT DWP (including within the anchorage area) will be held responsible for all cleanup costs. Vessels visiting the SPOT DWP are required to abide by the following regulation:

• 33 CFR § 155.370 (Oily Mixture [Bilge Slops]/Fuel Oil Tank Ballast Water Discharges on Oceangoing Ships of 10,000 Gross Tons and Above and Oceangoing Ships of 400 Gross Tons and Above That Carry Ballast Water in Their Fuel Oil Tanks);

VLCCs and other crude oil carriers will manage ballast water in accordance with International Maritime Organization (IMO) Standards. These standards include having a ship-specific ballast water management plan, carrying a record book, and exchange of water mid-ocean or with an on-board ballast water treatment system (IMO 2019b).

Entanglement Mitigation

During construction, anchor handling tugs will support the pipeline installation barge and will use a minimum of two stem anchors, two bow anchors, and four breast anchors. A deadman anchor will also be used during pipeline installation in the GoM. Anchors will be set and raised repeatedly during the installation of the two subsea pipelines. The SPM system will use fluke anchors and anchor chains to secure the buoy in position. The two SPM buoys will each be held in place by three fluke anchors and anchor chains, for a total of six anchors. The anchor chains would be equally spaced on a 1,043-foot radius circle with 1,080 ft of anchor chain between the anchor and the chain stopper on the buoy.

Table 2 provides the number of days anticipated for each phase of offshore construction.

Table 2. Time Period for Offshore Construction Activities

Facility Component	Number of Days Required for Installation
Pipe Laying	152
Jet Sledding	102
Platform Installation	65
PLEM and SPM Buoy Installation	88
Hydrostatic Testing	96

PLEM = pipeline end manifold; SPM = single-point mooring
Source: SPOT 2019a, Application, Volume IIa, Section I

Operations and Maintenance Procedures

All aspects of SPOT DWP safety and operations will be addressed in the Port Operations Manual (OPSMAN), which will require USCG approval prior to initiation of SPOT DWP operations as a condition of the license. The SPOT DWP will operate in accordance with the conditions of the license, the DWP regulations (33 CFR Subchapter NN), and the OPSMAN. After commencement of operations, the SPOT DWP operator(s) will be required to continuously assess operational performance and conditions under which routine and emergency actions are conducted, and incorporate lessons learned into the OPSMAN to enhance safety of personnel, infrastructure and the environment, and operations and safety.

As a condition of a DWP license, MARAD will require that the SPOT DWP develop a Prevention, Monitoring, and Maintenance Plan (PMMP) to describe procedures to be used during operations. Compliance with an approved PMMP will be made a condition of the license and will be incorporated as an Annex to the OPSMAN. The PMMP will satisfy the needs of Federal, State, and local agencies to ensure the prevention, monitoring and mitigation of the environmental impacts that may result from the construction and operation of the Port. The PMMP will address regulatory requirements and requirements of permits, approvals, and authorizations; project specific requirements; best management practices; and any other commitments made by the Applicant included in the application and Final EIS. The PMMP will provide Port personnel the necessary information, training, procedures, and equipment to implement the PMMP's requirements and integrate them into all Port operations. The Applicant will collectively work with Federal, State, and local agencies, as appropriate, to develop the PMMP. The PMMP will be regulatory and performance-based and include periodic evaluation of effectiveness to identify environmental protection improvements in the Port's operating area.

Following construction, SPOT will undertake a number of activities for startup and commissioning of the project, including dewatering, purging and packing, and purging and filling the pipelines and other project components. SPOT will first remove the hydrostatic test water from all components of the Project. Following dewatering, pigs, propelled by air, will be used to clean and dry the system. Nitrogen will then be pushed through the system to remove the air. Once the air is removed from the system, the nitrogen will be replaced by crude oil.

Crude oil will be transferred from onshore pipelines and terminal facilities. As VLCCs or other crude oil carriers moor at the SPOT DWP, the crude oil will be transferred from the land-based storage tanks to onshore pipelines, into the subsea pipelines, and to the offshore platform. The crude oil will be metered at the offshore platform and transferred through the 30-in-diameter pipelines to the PLEMs, through the underbuoy pipelines to the SPM buoys, and through the floating crude oil hoses to the VLCCs or other crude oil carriers for loading. SPOT anticipates the SPOT DWP would be called on by 365 VLCCs or other crude oil carriers per year (i.e., one per day). The loading time would be approximately 24 hours from VLCC or other crude oil carrier mooring to disconnection and transit back into the GoM. However, other crude oil carriers may not take as long to load as a VLCC due to their smaller capacities.

The SPOT DWP will be operated in accordance with the USCG-approved OPSMAN. The platform will be manned at all times. SPOT applicant will develop an operational spill response plan in accordance with OPA90 and Pipeline and Hazardous Material Administration's (PHMSA) implementing regulations in 49 CFR Part 194, the National Oil and Hazardous Substances Pollution Contingency Plan, applicable Area Contingency Plans, the USEPA Region 6 Regional Integrated Contingency Plan, and the One Gulf Plan. The plan will be developed to assist personnel with quickly, safely, and effectively responding to a crude oil spill either onshore or offshore, and will be prepared during construction of the proposed Project. This operational spill response plan will be developed in consideration of a worst case discharge .

Decommissioning

Decommissioning of the SPOT DWP will be performed when necessary or at the end of the useful life of the SPOT DWP Project. The anticipated life of the SPOT DWP Project is 30 years. Vessels and barges will be mobilized to remove offshore components of the SPOT DWP, which will be transported for disposal or recycling, or relocated to a Rigs-to-Reefs location. Any proposal to use decommissioned components of the SPOT DWP for artificial reef creation will require additional ESA Section 7 consultation at that time to assess the effects to ESA-listed species or designated critical habitat based on the details of the reef project. SPOT will prepare a decommissioning plan prior to conducting decommissioning activities. The main activities that will compose decommissioning of the SPOT DWP include:

- Removing navigational buoys and ancillary equipment;
- Disconnecting the floating crude oil loading and vapor recovery hoses;
- Removing SPM buoys and ancillary equipment, including the underbuoy crude oil and vapor recovery pipelines;
- Removing the PLEMs and crude oil and vapor recovery lines connecting to the platform and connecting to the subsea pipelines;
- Abandoning the subsea crude oil pipelines in place; and
- Removing the offshore platform.

SPOT will remove all lighted buoys, their chains, and their anchor blocks for transport onshore for disposal. The floating crude oil and vapor recovery hoses and underbuoy crude oil and vapor recovery pipelines will be flushed with seawater from the open end to the PLEMs. The floating hoses will be disconnected from the SPM buoys, capped by divers, and transported to shore for disposal. The underbuoy lines and SPM buoy anchor chains will be disconnected from the buoys and the fluke anchors. Fluke anchors will also be removed if the anchors are less than 15 ft below the natural bottom elevation. If the fluke anchors are more than 15 ft below the natural bottom elevation, the anchors will be left in place. The buoys, chains, and anchors will be transported to shore for disposal. The crude oil loading and vapor recovery pipelines between the platform and the PLEMs will be flushed from the platform to the PLEMs and back again using the pigging system. Any crude oil will be pushed into the subsea pipelines to shore. Following removal of the crude oil, flushing water and chemicals will be run through the lines. The discharge water will be separated until no hydrocarbons are identified in the flushing water. All free liquids on the platform, including the discharge water, will be drained from piping and vessels into an industry-approved transport container for transport to shore. The tie-in spools at either end of the lines will be removed and transported to shore for disposal. The pipeline ends will be capped, lowered to 3 ft below the natural bottom elevation, and covered with concrete mattresses. The PLEMs will be removed and transported to shore for disposal. The anchor piles will be cut approximately 15 ft below the natural bottom elevation and the upper pile sections removed for disposal.

To fully decommission the platform, any piping not cleaned via pigging the crude oil pipelines and hoses will be cleaned of hazardous chemicals, diesel, or other materials. Equipment onboard the platform, such as the cranes, will be tied down. Other components of the platform, such as the living quarters, main components of the main deck, and the helideck will be removed with a derrick barge and transported to shore for disposal. Deck legs will be cut at the top of the jacket and each of the decks will be lifted onto a cargo barge for transport to shore and disposal. Jacket piles will be cut from the inside approximately 15 ft below the natural bottom elevation via abrasive cutters, water cutters, or explosives. If explosives are used, that activity will require additional ESA Section 7 consultation at that time, based on the details of the removal project. The piles will be removed and transported to shore for disposal.

The subsea crude oil pipelines from the platform to state-owned submerged lands boundary (9 nmi from shore) will be abandoned in place. First, crude oil will be pushed from the offshore platform to the Oyster Creek Terminal using the pigging system, then the lines will be flushed with seawater in a loop until no hydrocarbons are identified in the flushing water. The tie-in spools at the platform end of the pipelines will be removed and transported to shore for disposal. The pipeline ends will then be capped and the lines lowered to 3 ft below the natural bottom elevation, then covered with concrete mattresses.

The portions of subsea pipelines located on state-owned submerged lands and installed via jet trenching, will be removed from the seafloor. After the pipeline is cleaned, a section of pipe near the HDD exit pit will be uncovered, cut, and a 40-ft pipe section will be removed. A plug will be installed in the end of the HDD segment and a concrete mattress would be installed to cover the pipe end. Cross bars will also be installed in the end of the offshore pipe segment. A vessel will then move to Galveston Area Block 463 and locate the pipe end, remove concrete mattresses, remove the blind flange, install a pig launcher, and run a pig with air from the launcher to the north end of the pipe segment near the HDD exit pit. The pipe will then be ready to be pulled through the stringer, onto the pipelay barge, and up the pipelay ramp to the bow of the barge.

The pipelay barge will remove the field joint and cut the pipe into 40-foot pieces, backing up to the next segment until about 64,800 ft of pipe is removed. The segments of pipe that remain will be completely filled with seawater, the laydown head will be trenched to a minimum of 3 ft below natural bottom, and a concrete mattress will be placed over the trenched laydown head end. The process will be repeated for both subsea pipelines. All recovered pipe joints will be transported to an onshore facility for disposal.

Where subsea pipelines will be installed on state-owned submerged lands via HDD construction methods, SPOT will comply with requirements of the Texas General Land Office Miscellaneous Easements contract, which may allow for pipeline segments installed via the HDD method to be abandoned in place, provided they are sufficiently buried to industry standards and pose no threat to human health and safety or to the environment.

The service vessel moorings will be among the last offshore components to be removed during decommissioning because they will remain in use during the decommissioning process for vessels performing certain activities. The mooring buoys will each be lifted onto the deck of a vessel and the anchor line disconnected. The anchor blocks will then be removed using a crane and transported to shore for disposal. If necessary, jetting may be used to loosen the anchor blocks from the seafloor.

Following decommissioning of all offshore components, a survey will be conducted to confirm no debris is left behind and SPOT will submit final reports to appropriate regulatory agencies.

Construction Conditions

SPOT has agreed to adhere to NMFS Southeast Region's Protected Species Construction Conditions (https://media.fisheries.noaa.gov/2021-06/Protected_Species_Construction_Conditions_1.pdf?null), and NMFS Southeast Region's Vessel Strike Avoidance Measures (https://media.fisheries.noaa.gov/2021-06/Vessel_Strike_Avoidance_Measures.pdf?null) throughout all in-water construction and decommissioning activities.

2.1.2 Texas GulfLink Deepwater Port Project

Texas GulfLink Holdings (hereafter referred to as TGL), a Subsidiary of Sentinel Midstream, LLC proposes to construct, own, operate, and eventually decommission a deep-water port (DWP) facility, which will load domestically-produced crude oil onto deep draft crude carrying vessels for export overseas. Use of the proposed TGL DWP will include the loading of West Texas Intermediate (WTI) and WTI-Light crude oil into tankers at flow rates of up to 85,000 barrels per hour (bph). Approximately 95 percent of the tankers to load at the proposed TGL DWP would be very large crude carrier (VLCC) vessels, which can carry cargos of approximately 2 million barrels of oil, with Suez Max Tankers making up the remaining 5 percent. Using an average loading rate of 65,000 bph (or approximately 33 hrs to load a VLCC [or equivalent volume]), approximately 12 tankers could be loaded per month (144 per year) from the proposed TGL DWP (i.e., more than 1 million barrels of oil per day). This frequency of tankers would allow for deliveries to storage tanks and a 42-hr port turnaround for each tanker.

TGL seeks to develop the DWP in the Houston/Freeport area to access the increasing crude oil supply provided by new pipeline projects from Cushing and West Texas to the Gulf Coast, and to take advantage of the offshore topography near Freeport, which can accommodate loading of deep draft VLCCs.

The proposed action has both onshore and offshore components. The onshore components will be located in Brazoria County, Texas, and will include a new Shoreside Support Facility at Port Freeport Public Docks, as well as a new 36-inch pipeline beginning at an existing 40-inch pipeline within the property of the Strategic Petroleum Reserve (SPR), and running to the proposed TGL Jones Creek Terminal (incoming pipeline) for approximately 9 miles. The TGL Jones Creek Terminal will be an onshore storage tank farm that will store the crude oil in up to 12 above-ground floating roof storage tanks. An administration building will be constructed on site to act as the operational headquarters for the Jones Creek Terminal. From the TGL Jones Creek Terminal, an outgoing onshore pipeline (42-inch pipeline approximately 12.1 miles in length) will be constructed to connect with the 42-inch offshore pipeline just southwest of the Brazos River delta. From the connection with the outgoing onshore pipeline, the new 42-in offshore pipeline will run approximately 32.3 miles out to the proposed TGL DWP. Offshore components of the proposed TGL DWP will be sited at a location with a water depth of approximately 104 feet within the Galveston Area Lease Blocks GA-423 and GA A-36 (Figure 8). Offshore components will consist of one fixed platform and two single-point mooring (SPM) buoy systems, along with various offshore support vessels (OSV), including one OSV equipped with a Vapor Processing Module (VPM) for capturing vaporized volatile organic compounds during loading of the VLCCs.

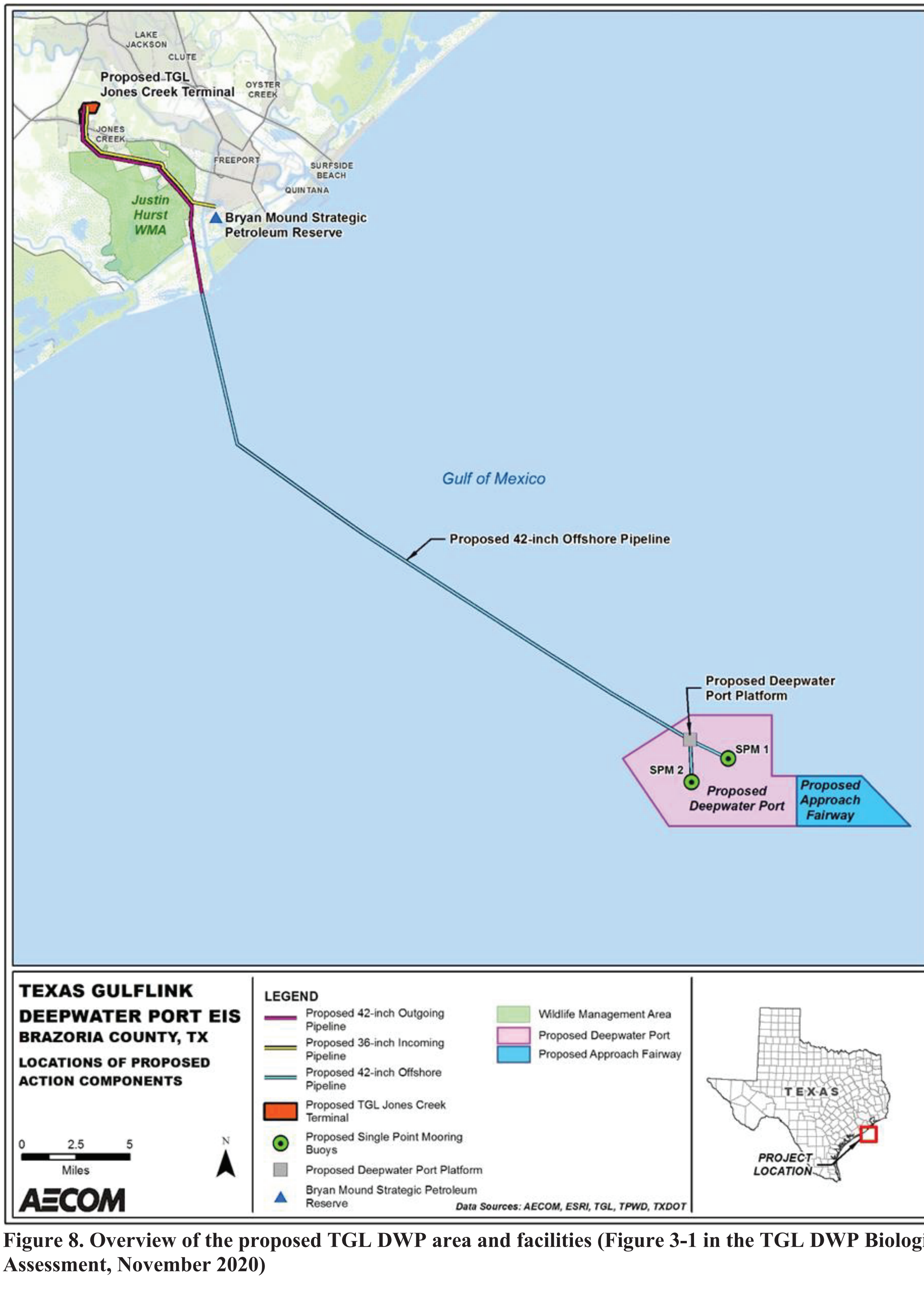


Figure 8. Overview of the proposed TGL DWP area and facilities (Figure 3-1 in the TGL DWP Biological Assessment, November 2020)

The proposed Shoreside Support Facility at the Port Freeport Public Docks will have a warehouse to provide storage space, as well as loading and unloading facilities for support vessels for offshore construction and operations. TGL will contract a marine vessel operator and a marine support vendor to provide these services. Therefore, the TGL DWP project does not include construction or additional utilities (i.e., water, power, or communication) for the Shoreside Support Facility.

Pipeline Description and Installation

Horizontal Directional Drilling (HDD)

The sections of both the incoming and the outgoing pipelines beneath Durazno Plantation, Jones Creek, and two wetlands will be installed using the HDD method. As described above under the SPOT DWP project, HDD is a trenchless method of installing pipelines, which often requires temporary, linear work are for constructing the pipeline before it is pulling through the boring.

TGL has developed a Proposed HDD Execution Plan (Appendix K of the TGL DEIS) that defines the technical procedures to be used, as well as safety, security, and environmental protection procedures to be followed. The HDD process includes use of a drilling rig and other heavy equipment which use petroleum fuels. In addition, the process requires use of drilling fluids composed primarily of water and bentonite clay in the borehole, and also generates drill cuttings that would require disposal. To address these items, the environmental protection measures included in the HDD Execution Plan include secondary containment for stored hazardous materials, emergency response procedures in the case of spills or releases of hazardous materials or drilling fluids, and offsite disposal of drill cuttings at an authorized disposal facility.

Direct Pipe Installation (DPI)

DPI is similar to HDD in that it is trenchless. The DPI method uses a specialized drilling head that is attached to a section of the pipe. The control systems for the drilling head run through the pipe and the pipe is pushed forward as the drilling head cuts the hole. As the drilling head progresses and the pipe advances, additional pipeline sections are welded onto the open end. TGL proposes to use DPI to install pipeline sections beneath the Brazos River, the Gulf Intracoastal Waterway (GIWW), and the shore approach. For the direct pipe boring that will emerge approximately 1,500 feet offshore, a spud barge or anchored barge equipped with a crane operating a dredge bucket will dig an exit pit at the exit point. Spoils from the pit will be spread along the sea bottom within the construction right-of-way (ROW). The exit hole will be located at a water depth of 8 feet. The end of the section of pipeline will be recovered and pulled into the welding stalls on the shallow water lay vessel. The drilling head will be removed from the end of the pipeline. The concrete-coated (offshore outgoing) pipeline will be connected and welded to the section of pipe and lowered to the sea floor.

At trenchless entry and exit points, a larger additional temporary workspace will be needed to accommodate the necessary equipment. This would be the case at the HDD entrance on the west side of the Brazos River, where the pipeline route leading into the HDD section is not straight. At this location, a linear additional temporary workspace of approximately 3,000 feet will be needed to provide laydown area for the pipeline to be placed into the HDD. Because of a curve in the pipeline route at this location, that workspace will not be coincident with the permanent pipeline ROW.

Because there is no roadway access for vehicles and equipment on Wolf Island between the Intracoastal Waterway and the shoreline, equipment needed to install the pipeline across the island and to install the HDD underneath the shoreline will be brought to the island by barge.

Offshore Pipeline Construction

The proposed offshore 42-inch pipeline will traverse 28.1 nautical miles southeast from the outgoing pipeline direct pipe boring exit point to the proposed TGL DWP at lease block GA-463. At the proposed TGL DWP, the pipeline will enter the fixed offshore platform, and will split into two pipelines, each approximately 1.25 nautical miles in length. These two pipelines will connect to two SPM buoys where crude oil will be loaded onto the tankers.

A shallow water conventional anchored pipelay vessel will be positioned at the exit of the direct pipe boring and used to recover the end of the pipeline. The concrete-coated offshore pipeline will be connected and welded to the pipe and lowered to the sea floor. From there, laying of the offshore pipeline will proceed approximately 28.1 nautical miles to the proposed fixed offshore platform. The pipeline located beyond the 40- to 60-foot water depth will require a larger anchored or dynamic positioning pipelay vessel, depending on the back-tension installation requirements.

After each new joint of pipe is welded, the pipelay barge will move forward by pulling in on the bow anchor line while releasing cable to the stern anchor line. The lateral anchor wires will keep the pipelay barge restrained laterally. The actual touchdown of the pipeline on the sea bottom will be verified by a remotely operated vehicle (ROV) with ultra-short baseline positioning equipment to provide real-time positioning. The feedback from the ROV will allow the pipelay barge to make adjustments as required to center the pipeline within the ROW.

Trenching will be used to bury the offshore pipeline so that the top of pipe is a minimum of 3 feet below the mudline and a minimum of 10 feet below the mudline in the shipping fairway. The offshore pipeline will be protected from corrosion by an anti-corrosion coating and galvanic sacrificial anodes and will have concrete weight for ballast.

In most areas, a sled jet trenching system will be used to excavate a trench and lower the pipeline to the required cover depth. Several passes may be required to lower the pipeline to the required depth to meet the cover requirements. After the pipelines have been placed and covered, a post-burial survey will be performed to verify that the required protective cover over the pipeline is present.

Construction of the pipelines from the proposed fixed offshore platform to each SPM buoy will commence with an initiation cable connected to the platform. The end of the SPM buoy pipeline will be installed with a flange connected to a lay down head. Approximately 1.25 nautical miles of the 42-inch pipeline will be laid from the platform to each SPM buoy. The pipeline will be lowered onto the sea bottom with the winch cable. The end of the pipeline will be verified using subsea survey positioning.

To ensure safe and efficient anchor-handling operations during construction of the pipeline, construction crews and vessels will adhere to TGL's Anchor Handling and Mooring Procedures guidance document (Abadie-Williams, 2021). These procedures are designed to ensure the safety of personnel and protect underwater resources, equipment, and existing underwater facilities, such as unburied existing pipelines, power cables, wells, subsea valves, and other structures.

Offshore Platform Construction

The proposed offshore platform will consist of the platform jacket, deck, four pilings, appurtenances and topsides, as well as three 42-inch risers. The platform is a 4-leg fixed platform with three (3) jacket horizontal framing elevations and supported at the seabed by means of piling driven through un-grouted legs. The jacket supports a four-level topside including Main, Meter, Cellar, and Sump decks (Figure 9). The jacket risers and lift rigging will be pre-installed prior to delivery of the jacket to the site. The jacket (a steel frame supporting the deck and the topsides) will be loaded onto a material barge for transport to the proposed TGL DWP. The platform may be installed before or after the completion of the proposed offshore pipeline.

Prior to platform construction, the sea bottom will be inspected for debris and a derrick barge will be mobilized to the proposed TGL DWP area. The jacket will be lifted and set on the sea bottom to the correct orientation. Once on the sea bottom, the jacket levelness and attitude will be recorded to determine the pile stabbing sequence.

The first pile section will be long enough to penetrate the sea floor and will become self-supporting with the additional penetration driven with a hydraulic hammer to the top of the jacket. Subsequent pile sections will be lifted and stabbed onto the previous pile sections with preinstalled stabbing guides on the bottom of the add-on pile sections. Approximately 8 hours of welding will be required to securely attach each pile section to the previously driven section, resulting in relatively short periods of impact hammering (approximately 200-700 strikes, depending on which substrate layer is being penetrated), followed by longer interludes of welding. The final pile sections will be secured to the top of the jacket leg using a crown type shim.

The topsides and living quarters associated with the proposed fixed offshore platform will be loaded on a separate material barge and sent to the platform location while the shims are installed on the jacket legs. The topsides would be lifted and set on top of the pile cut sections. The bottom of the deck legs will have stabbing guides that are stabbed into the pile sections at the top of the jacket. Once an adequate amount of weld material has been installed on the deck leg and pile splice, the fabricated living quarters package would be lifted and set onto the platform. The living quarters package will be secured by welding it to the platform topside framing.

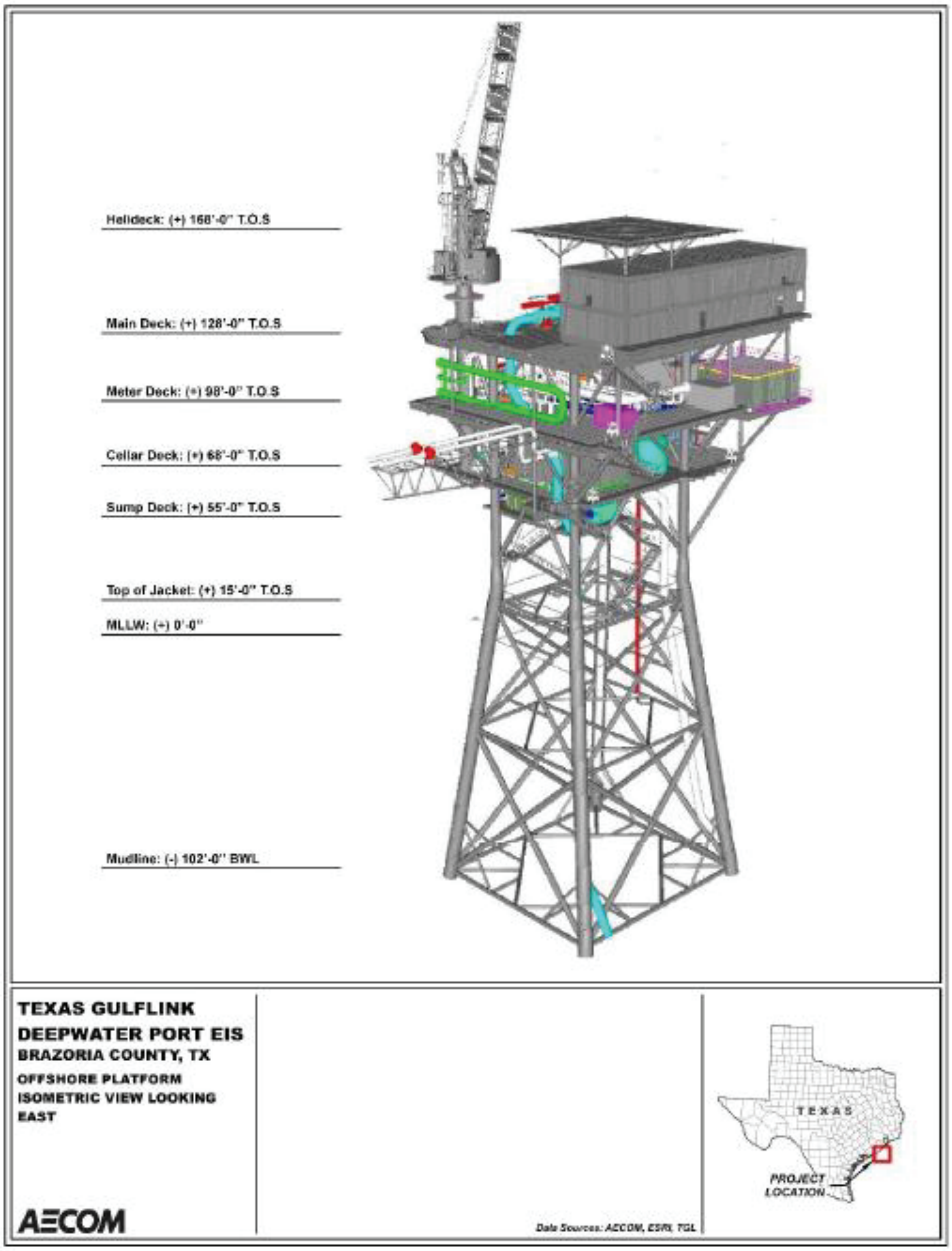


Figure 9 Schematic drawing of proposed TGL offshore platform (Figure 2.2-5 in the TGL DEIS, November 2020)

The proposed fixed offshore platform will be supported by four 66-in-diameter steel piles, which are larger than anchor-type piles. The 66-in diameter piles will be installed using a conventional impact hammer operating off a derrick barge, following the installation process described above. Additional anchor-type, steel piles will be installed for the 2 pipeline end manifolds (PLEMs) (four 24-in piles each) and 2 SPM buoys (nine 54-in piles each). The platform and SPM buoy piles will be installed by impact pile driver, and the PLEM piles will be installed by vibratory pile driver. A total of 4 days will be needed to complete the installation of all 4 offshore platform piles (1 pile per day), 2 days will be needed to complete the installation of all 8 PLEM piles, and 6 days will be needed to complete the installation of all 18 SPM buoy anchor piles. Thus, the total duration of pile-driving activities will be approximately 12 days. Table 3 summarizes the pile-driving information for the 3 types of structures. This information is used in the modeling of noise-related impacts to ESA-listed species discussed below.

Table 3. Pile-Driving Details for Each of the 3 Types of Structures

Parameter	**Piles Supporting the Offshore Platform**	**Piles Supporting the SPM Anchors**	**Piles Supporting the PLEMs**
Pile type	steel pipe	steel pipe	steel pipe
Pile number	4	18 (9 per SPM)	8 (4 per PLEM)
Pile diameter	66 inches	54 inches	24 inches
Pile-driving method	impact	impact	vibratory
Number of strikes (impact) or seconds (vibratory) per pile	1,261 strikes	254 strikes	1,200 seconds
Driving time per pile	12 hrs	1 hr	20 minutes
Number of piles per day	1	3	4
Number of driving days	4	6	2
Attenuation device	bubble curtain	none	none
Attenuation rate	5 dB	none	none

Bubble Curtain Design

TGL proposes to install two sets of bubble curtain tube rings on the sea floor around the base of the jacket legs to minimize the noise energy generated into the water column during impact pile driving activities. Each set will consist of two tube rings encircling two of the jacket legs (Figure 10). A support vessel will be outfitted with an air compressor to provide 800 cubic feet per minute of air compression capacity to feed air to the bubble tubes during pile installation (Figure 11).

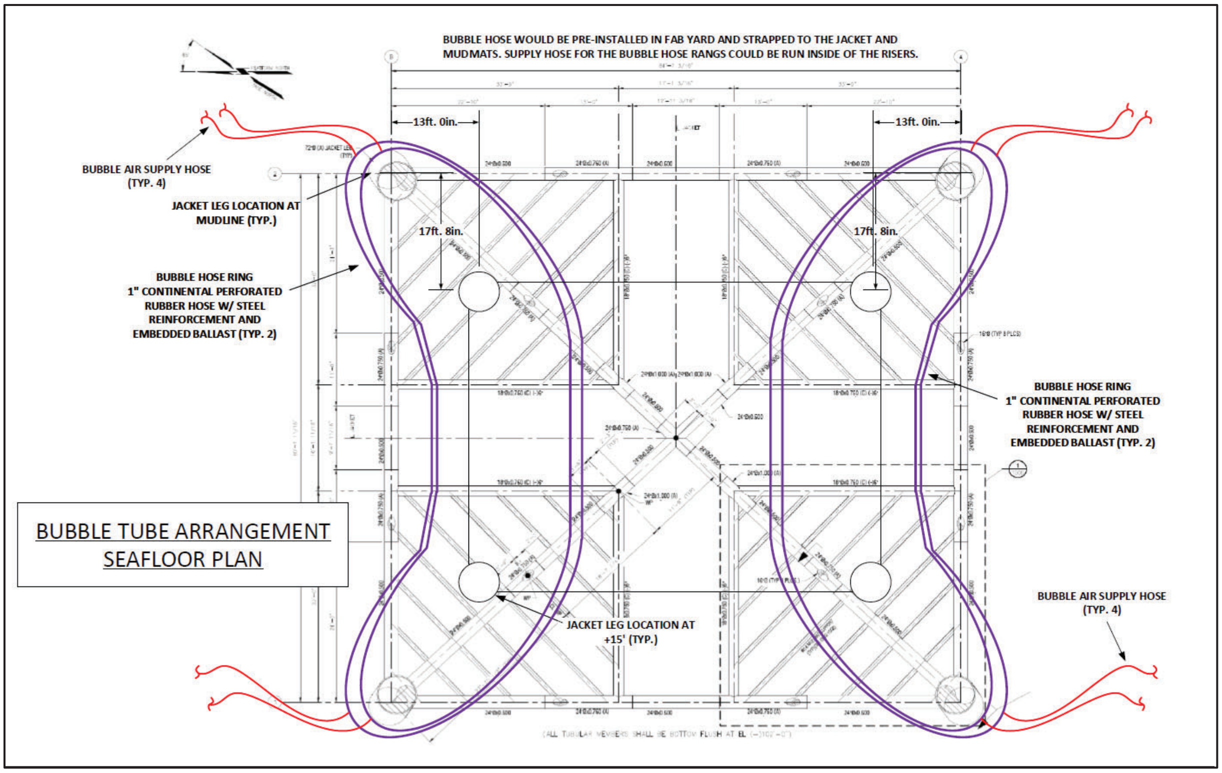


Figure 10. Schematic drawing of proposed bubble curtain design, plan view (drawing provided by Abadie Engineering LLC, April 2021)

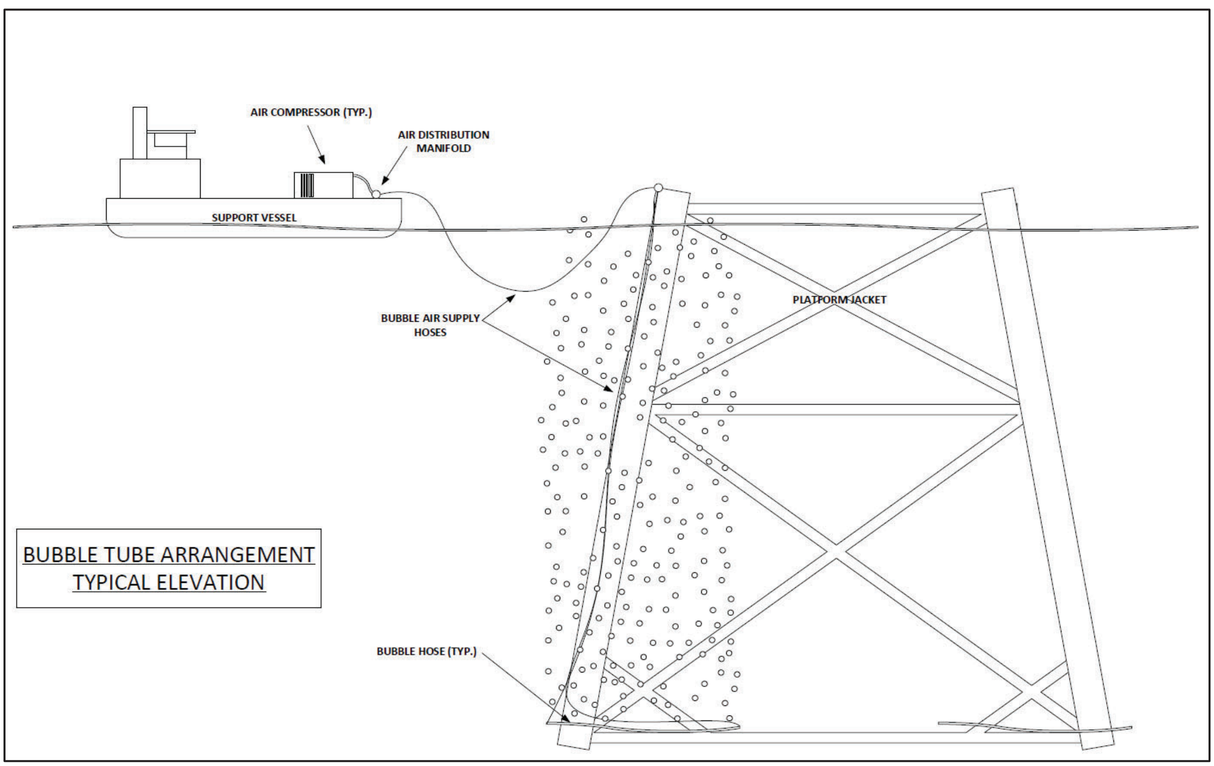


Figure 11. Schematic drawing of proposed bubble curtain design, side view (drawing provided by Abadie Engineering LLC, April 2021)

The proposed action includes a Vapor Processing Module (VPM) mounted on a third party offshore support vessel (OSV) to be chartered for supporting the DWP. The VPM would process vapor emissions, while the OSV will maintain its station alongside the VLCC using dynamic positioning thrusters. The OSV will remain 100 feet off the side of the tanker with a single 16-inch diameter vapor hose connected to the VLCC during cargo loading. The vapor hose will be protected by the VLCC and OSV during cargo loading. The vapor hose will be purged with nitrogen back to the VLCC tank after each loading. Disconnecting the ends of the vapor hose between consecutive loads will allow for proper drainage of any liquids that may condense in the hose.

The VPM will compress, condense, and cool the vapors into two streams: (i) surplus volatile organic compounds (VOCs) and (ii) liquid VOCs. Condensing recovered vapors into liquid VOCs requires the intake of seawater for cooling. The maximum cooling water intake that the OSV could use based on design capacity will be 980 cubic meters per hour. The actual intake will be much lower because cooling is only needed during active cargo loading. Onboard the OSV, the surplus VOCs and liquid SOVs will be consumed as fuel for the OSV's gas turbines. The liquid VOCs will also be stored as a liquid in the OSV deck tanks.. No venting of vapors will occur on the OSV while loading. Excess liquid VOCs will be offloaded ashore at Port Freeport. The OSV will be required to make the trip to shore after 2.5 VLCC loadings to offload excess liquid VOC cargo, or about once per week.

Offshore Hydrostatic Testing

Offshore Hydrostatic Testing will be conducted on the proposed fixed offshore platform components, the offshore pipelines between the platform and the SPM buoys, and the SPM buoys. The offshore hydrostatic testing will include a separate test for each of the SPM buoy pipelines. Seawater from the offshore intake will be used to perform these tests.

Following testing, the seawater will be discharged from Outfall 005 at the proposed fixed offshore platform. The seawater remaining in the pipeline between the proposed TGL Jones Creek Terminal and the platform will also be discharged through Outfall 005 at the platform as the first cargo of crude oil is pumped into the pipeline at the terminal. The seawater discharge will be directed in an upward path from the seafloor in order to avoid the potential for scour. All tested components will be new and unused, with no potential for contaminating the water.

Discharged water will also be subjected to visual inspection and, depending on National Pollutant Discharge Elimination System (NPDES) permit requirements, possible water quality testing before discharge. In addition to hydrostatic testing prior to the beginning of operations, ad-hoc hydrostatic testing events may occur throughout the operations period. These testing events are also addressed in the NPDES permit application.

Offshore Operational Water Intake and Discharges

The proposed TGL DWP offshore platform will be designed and constructed to manage discharge of stormwater and treated sanitary wastewater. The platform's saltwater intake will have a flow rate capacity of 100 gpm for the purposes of maintaining pressure in the firefighting system. The intake pump will have a suction caisson extending down roughly 5 feet below the pump entrance. The approach velocity at the screened entrance to the caisson will be 0.6 feet/second, to allow fish and other mobile organisms to avoid entrainment and impingement.

Potable and sanitary water for operation of the TGL DWP will be obtained in Freeport and supplied to the offshore platform by support vessels on a weekly basis. Seawater used for the offshore platform operations will be limited to testing the firewater pump for 2 hours per month, at a rate of 12,000 gpd. The remainder of the use of seawater will be associated with the operation of VLCCs and the vapor recovery vessel.

VLCCs will withdraw seawater for engine cooling, and discharge ballast water, while moored at the TGL DWP. All tankers loading at the TGL DWP will be required to follow their own approved Ballast Water Management System. The tankers will also be required to use approved equipment and follow and maintain records for ballast water and other operational discharges (e.g., bilge, sanitary discharges) that are compliant with the International Convention for the Prevention of Pollution of Ships (MARPOL) and USCG standards. Most VLCCs are registered under the Ballast Water Convention and also would manage ballast water in accordance with IMO Standards. These standards include having a specific ballast water management plan, documentation in a record book, and exchange of water mid-ocean or with an on-board ballast water treatment system (IMO 2019). VLCCs that manage ballast water within the GoM will treat the water using physical or chemical treatment, or a combination of the two, to remove invasive species.

Discharges from the proposed fixed offshore platform will include sanitary wastewater, stormwater from deck drainage, and fire water pump test discharge.

Operations and Maintenance Procedures

Because the proposed TGL DWP project includes both onshore and offshore components, its design and operations are regulated by more than one Federal agency. The required contents of the TGL Operations Manual, which specifies the design and operating requirements for DWPs, are provided in 33 CFR §150.15. TGL has developed a (proprietary) Operations Manual which, in turn, includes a Safety Manual, Facility Security Plan (FSP)/Deepwater Port Security Plan (DWPSP), Facility Response Plan (FRP), and Garbage Management Plan. In addition, as a condition of a DWP license, MARAD will require that TGL develop a Preventative Monitoring and Maintenance Plan to describe procedures to be used during operations.

Separately, design and operations of pipelines that transport hazardous liquids, including pipelines on the Outer Continental Shelf (OCS), are regulated by the U.S. Department of Transportation’s Pipeline and Hazardous Safety Administration (PHMSA). The required contents of a procedural manual, which specifies the operating and maintenance requirements for pipelines that transport hazardous liquids, are specified in 49 CFR § 195.402. TGL has developed a Hazardous Liquid Pipeline Operations, Maintenance, and Emergency Response Manual which addresses the requirements of the procedural manual in 49 CFR § 195.402, including the requirement for an Emergency Response Plan.

Activities associated with onshore and offshore, planned and unplanned maintenance would be similar to the activities described for onshore and offshore construction. Right-of-way maintenance will include ROW and mainline valve inspections. Unplanned or emergency repairs that require site disturbance to expose buried pipeline may be needed. Procedures for unscheduled ("non-routine") repairs, regardless of whether they require site disturbance, will be addressed in the Hazardous Liquid Pipeline Operations, Maintenance, and Emergency Response Manual and Preventative Monitoring and Maintenance Plan. If needed, pipeline repair plans will be developed and submitted to applicable agencies.

Routine maintenance during operations will include visual inspections, as well as cleaning and inspection of the inside of the pipelines. Visual inspections of the proposed TGL Jones Creek Terminal will include the tank shells, roofs, seals, valves, and other general equipment. Pigs will be used to inspect and clean the interior of the pipelines. The terminal will be equipped with a sump system to manage small quantities of crude oil drained from pig launchers and receivers and from crude oil piping while it is being maintained. The sump system will return the small quantities of crude oil to one of the main storage tanks, where it will ultimately be pumped into the outgoing pipeline for delivery to the proposed TGL DWP. Blast residue and water residuals will be contained and disposed in accordance with Federal, State, and local regulations.

To reduce the risk of a vessel strike during the TGL DWP construction, all construction vessels will comply with NOAA's NMFS Southeast Region's Protected Species Construction Conditions (NMFS 2021). These conditions require, among other things, that operations of moving equipment shall cease if a protected species is observed within 150 feet of operations and that activities shall not resume until the protected species has departed the project area of its own volition (e.g., species was observed departing or 20 minutes have passed since the animal was last seen in the area). Construction vessels will also comply with NOAA Fisheries' Vessel Strike Avoidance Measures (NMFS 2021). These measures require, among other things, that vessel operators and crew maintain a vigilant watch for protected species (i.e., marine mammals and sea turtles) to avoid striking sighted protected species. In addition, these measures also require all vessels operate at "idle/no wake" speeds at all times while in the project construction area, while in water depths where the draft of the vessel provides less than four feet of clearance from the bottom, and in all depths after a protected species has been observed in and has recently departed the area.

Decommissioning

During decommissioning, the proposed TGL Jones Creek Terminal property will be restored to its original condition. All improved surfaces except the Administration Building and associated utilities, including piping, tanks, and equipment, and facilities, will be removed. Buried pipes and valves will be excavated and removed. Structure foundations will also be removed. Deeper foundations, such as grout columns, will be removed to depth, and the remainder will be covered and abandoned in place. The ground surfaces would be graded and re-vegetated to original condition. The Administration Building and associated utilities would be maintained in place.

The buried onshore pipeline segments will either be abandoned in place or removed, depending on the agreement with the landowner. Where the pipeline is to be abandoned in place, the pipeline will undergo a repeated pigging and flushing process to remove all hydrocarbons and solids. After the final flushing, the pipeline will be filled either with nitrogen or with water treated with corrosion inhibitor and biocides to prevent deterioration. Where the pipeline comes to the surface, the above ground portion of the pipeline will be removed to 4 feet below the ground surface. The remaining ends of the buried pipeline will be capped. In areas where pipeline removal is required, the pipeline will be exposed by trenching and removed. The land surface will be backfilled and revegetated according to agreements with the landowner.

TGL has developed detailed plans for decommissioning the offshore TGL DWP as part of the GA 423 Decommissioning Estimation Study (TSB Offshore 2019). The GA 423 Decommissioning Estimation Study describes the methodology and estimates costs for two potential options for decommissioning some project components, and therefore does not propose one or the other method at this time. For purposes of analysis in this Opinion, both proposed methods will be evaluated as decommissioning methods as part of the proposed TGL DWP project. The two methods are summarized in Table 4.

Table 4. Option 1 and 2 Decommissioning Methods for Offshore Components

Component	Option 1 Method	Option 2 Method
Offshore Pipeline – Texas State Waters	Abandon in place	Complete removal
Offshore Pipeline – Federal Waters	Abandon in place	Abandon in place
Platform	Complete removal	Leave portions in place for partial reef*
Pipeline – Platform to PLEMs	Abandon in place	Abandon in place
PLEMs, SPMs, and Floating Hoses	Complete removal	Complete removal
Options 1 and 2 are not mutually exclusive. The ultimate decommissioning method may include Option 1 for one component, and Option 2 for another component. * Any proposal to use decommissioned components of the TGL DWP for artificial reef creation will require additional ESA Section 7 consultation at that time, to assess the effects to ESA-listed species or designated critical habitat based on the details of the reef project.		

In general, TGL plants to completely remove the PLEMs, SPMs, and floating hoses, and to abandoned in place the pipelines in Federal waters in accordance with requirements of 30 CFR Part 250. For the PLEMS, SPMs, and floating hoses, crews on an anchor handling tug and a dive support vessel will dismantle all hose connections, sever all mooring chains, and remove the remaining floating components.

The pipelines to be abandoned in place will be pigged with 100 percent of the pipeline volume using seawater. The seawater and relict product would be treated through a series of carbon filters. The hydrocarbons will be captured in tanks aboard a work boat, and the treated water will be discharged. Once a sheen test has been approved, the segments above the seafloor and buried segments will be removed to a distance of approximately 20 feet from the PLEM. There the pipeline will be cut by divers, and the remaining open end capped and buried 3 feet below the mud line.

If removal of the 8.7 nautical mile segment of the pipeline in Texas state waters is required, the pipeline would be purged using the same procedures described above. A Semisubmersible

Dredging Pump would be used to expose the buried pipeline using a jetting process. Divers would then cut the pipeline into 80-ft long sections. The sections would be lifted by crane and placed onto a pipelay barge. The removed pipe would be disposed of at an onshore scrap yard.

The first phase of decommissioning of the platform will be preparation. All piping and equipment that contained hydrocarbons will be flushed. All equipment that is to be removed separately from the deck will be cut free using oxygen-acetylene torches. Lifting eyes needed to connect slings for removal will be attached. Once preparation is complete and removal crews are mobilized, the platform decks will be cut free from the platform legs with oxygen-acetylene torches. Slings to be used to lift the deck will be connected to the lifting eyes. A crane will then lift the deck and place it onto a cargo barge. In the open legs of the piles, a jetting process will be used to remove mud to a depth of 15 feet below the mud line. Then, an abrasive tool will be placed within the pile to the bottom and would sever the pile. The jacket will then be lifted by crane and placed on a cargo barge, transported to shore, and disposed as scrap. If explosives are used, that activity will require additional ESA Section 7 consultation at that time, based on the details of the removal project.

Depending on agency approvals at the time of decommissioning, the jackets may be partially removed. However, TGL notes that there is a requirement for 85 feet of clearance between the top of the jacket and the water line. Since the water depth is only 105 feet deep, complete removal would extend a maximum of 20 feet below mudline, and partial removal would only remove the jacket to a height of 20 feet above mudline. Therefore, the partial removal process would save removal of approximately 40 feet of jacket. Since partial removal results in other requirements for installing buoys, full removal may be more cost effective than partial removal.

Construction Conditions

The applicant has agreed to adhere to NMFS Southeast Region's Protected Species Construction Conditions (https://media.fisheries.noaa.gov/2021-06/Protected_Species_Construction_Conditions_1.pdf?null), and NMFS Southeast Region's Vessel Strike Avoidance Measures (https://media.fisheries.noaa.gov/2021-06/Vessel_Strike_Avoidance_Measures.pdf?null) throughout all in-water construction and decommissioning activities.

2.2 Action Area

The proposed SPOT DWP will be located in Federal waters on the continental shelf of the GoM, in Galveston Area OCS Lease Blocks 463 and A-59, approximately 27 mi off the coast of Brazoria County, Texas. The TGL DWP will be located the same distance from shore (27 mi), approximately 9 miles from the SPOT DWP, in the Galveston Area OCS Lease Blocks GA-423 and GA A-36 (**Error! Reference source not found.**). Bathymetry surveys identified slopes of approximately 12 ft per mile for the first few miles from shore, at which point the slope levels off to approximately 2 ft per mile for the rest of the distance out to the DWP sites. Water depth at the proposed SPOT DWP site is 115 ft, and water depth at the TGL site is 105 ft.

Figure 12. The SPOT deep water port location site (NAD83 is 28.466394, -95.123473) and TGL deep water port location (NAD83 is 28.55167, -95.02833) (©2022 Google)

The action area is defined by regulation as all areas to be affected by the Federal action and not merely the immediate area involved in the action (50 Code of Federal Regulations [CFR] 402.02). When looking at the effects of the proposed actions on ESA-listed species under NMFS's jurisdiction, the action area includes the pipeline routes and terminal sites, along with the broader areas affected by construction activities (such as turbidity from jet trenching activities, noise and sound pressure waves from pile driving, etc.), as well as the areas affected by operational activities (such as VLCC transit routes, potential oil spill areas, etc.). For example, turbidity generated from jet sledding activities along the SPOT DWP pipeline route would exceed background levels over a maximum area of about 19,044 acres, and sediment deposition would occur over a maximum area of about 6,210 acres. The largest area of potential effects from each project is based on modeling of potential oil spills that might result from project operations. The oil spill impact areas, along with the VLCC trans-Gulf shipping routes outside of the spill impact areas, overlap all other areas of project effects, and therefore define the overall boundary of the action area for the projects. These oil spill impact areas are described more thoroughly in Section 5 (Effects of the Action on Species and Critical Habitat). Figure 13 and **Error! Reference source not found.** below show the worst case impact areas for the SPOT and TGL projects (respectively) and Figure 15 shows the likely transit routes for the carriers utilizing the DWPs. Together, these figures show the overall action area for the projects. Due to the close proximity of the two projects, the similarity of project effects, and the overlapping oil spill impact areas, we will refer to the action area for the SPOT DWP and TGL DWP as a singular action area.

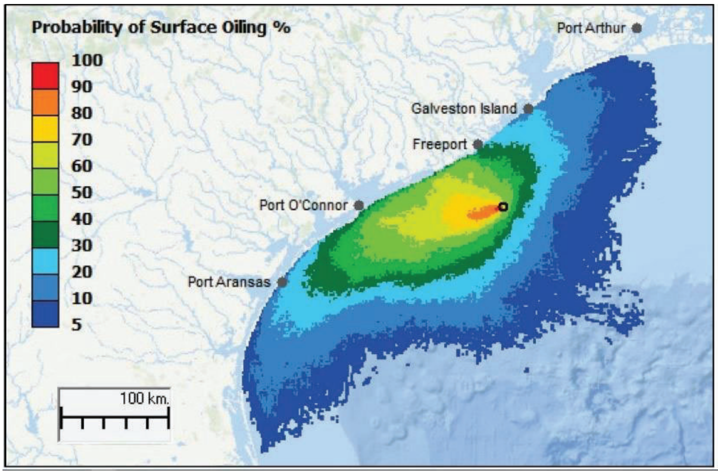


Figure 13. Worst-case impact area for SPOT DWP (Figure 4.6-13 in the SPOT FEIS, July 2022)

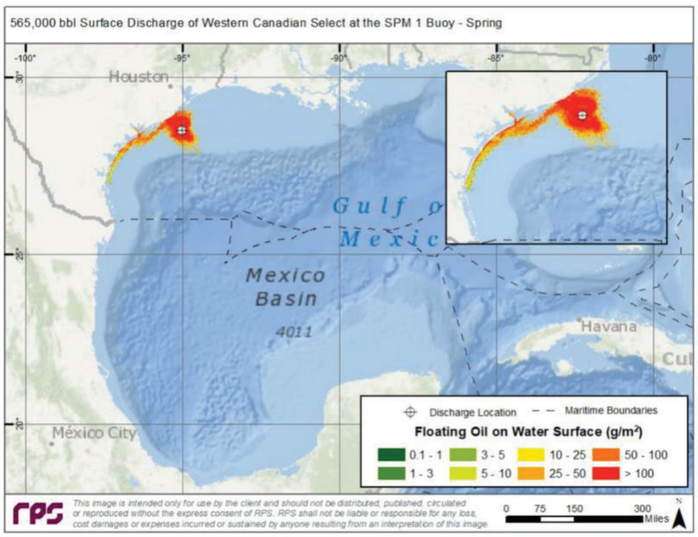


Figure 14. Worst-case impact area for TGL DWP (Figure 3-63 in Appendix I of TGL DEIS, October 2020)

Figure 15. Likely shipping lanes for carriers utilizing the DWPs (NMFS stock figure, updated May 2022)

Five species of seagrass are found along the Texas coast, along with floating *Sargassum* algal beds in open water areas offshore. The closest seagrass beds are located in Christmas Bay, more than 35 n mi from the proposed DWPs. Substrates in the action area consist of sandy sediments near the coastline that grade into silty clays and clays farther offshore.

Portions of the proposed offshore pipelines and DWPs would be located within three Marine Protected Areas (MPAs): Reef Fish Stressed Area, Reef Fish Longline and Buoy Gear Restricted Area, and Texas Shrimp Closure. The closest MPA sanctuary to the proposed DWPs is the Flower Garden Banks National Marine Sanctuary, which is about 87 mi southeast of the DWPs.

Once outside the influence of coastal anthropogenic processes and surface water runoff, water quality in the marine environment of the OCS typically improves (Kennicutt 2017b). Exceptions to this are waters just above naturally-occurring hydrocarbon (oil and gas) seeps, as well as localized and ephemeral effects on water quality due to the discharge of produced waters around oil and gas platforms (Kennicutt 2017a). OCS waters receive fewer pollutant discharges from human activities and experience greater mixing rates than coastal areas due to the large volume of receiving water (Kennicutt 2017b).

In the central GoM, hydrocarbon seeps are widespread and contribute hydrocarbons to the surface sediments and water column. The estimated total volume of oil seepage in the GoM exceeds 42 million gallons a year (Kennicutt 2017a). In addition to hydrocarbon seeps within the GoM, water quality can be affected by naturally occurring chemicals from methane seeps, hydrate mounds, and subsurface brines (Vigneron et al. 2017), which can contribute to the additional carbon load in deeper waters, and the observed decreased oxygen levels from enhanced chemical and biological oxygen demand.

3 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

This section identifies ESA-listed species and designated critical habitat under NMFS's jurisdiction that may occur in or near the action area and evaluates which of those may be affected by the proposed actions. Effects determinations are also summarized in Table 5. This section also describes the status of listed species and critical habitat that may be adversely affected by the proposed actions.

Table 5 provides the effect determinations for species the MARAD, USCG, and NMFS believe may be affected by the proposed actions. Please note abbreviations used in Table 5: E = endangered; T = threatened; NLAA = may affect, not likely to adversely affect; NE = no effect; N/A = not applicable

Table 5. Effects Determination(s) for Species the Action Agency Agencies and/or NMFS Believe May Be Affected by the Proposed Actions

Species	ESA Listing Status	Listing Rule/Date	Most Recent Recovery Plan/Outline Date	Action Agency Effect Determination	NMFS Effect Determination
Sea Turtles					
Green (North Atlantic DPS)	T	81 FR 20057/ April 6, 2016	October 1991	NLAA	LAA
Green (South Atlantic DPS)	T	81 FR 20057/ April 6, 2016	October 1991	NLAA	LAA
Kemp's ridley	E	35 FR 18319/ December 2, 1970	September 2011	NLAA	LAA
Leatherback	E	35 FR 8491/ June 2, 1970	April 1992	NLAA	LAA
Loggerhead (Northwest Atlantic DPS)	T	76 FR 58868/ September 22, 2011	December 2008	NLAA	LAA
Hawksbill	E	35 FR 8491/ June 2, 1970	December 1993	NLAA	LAA
Fish					
Smalltooth sawfish (U.S. DPS)	E	68 FR 15674/ April 1, 2003	January 2009	NLAA	NE
Gulf sturgeon (Atlantic sturgeon, Gulf subspecies)	T	56 FR 49653/ September 30, 1991	September 1995	NLAA	NE
Nassau grouper	T	81 FR 42268/ June 29, 2016	2018	NLAA	NE
Giant manta ray	T	83 FR 2916/ January 22, 2018	2019	NLAA	LAA
Oceanic whitetip shark	T	83 FR 4153/ January 30, 2018	2018	NLAA	LAA
Invertebrates and Marine Plant					
Elkhorn coral (*Acropora palmata*)	T	71 FR 26852/ May 9, 2006	March 2015	NE	NLAA
Boulder star coral (*Orbicella franksi*)	T	79 FR 53852/ September 10, 2014	N/A	NE	NLAA
Mountainous star coral (*Orbicella faveolata*)	T	79 FR 53852/ September 10, 2014	N/A	NE	NLAA
Lobed star coral (*Orbicella annularis*)	T	79 FR 53852/ September 10, 2014	N/A	NE	NLAA
Marine Mammals					
North Atlantic right whale	E	35 FR 18319/ December 2, 1970	June 2005	NLAA	NE

Species	ESA Listing Status	Listing Rule/Date	Most Recent Recovery Plan/Outline Date	Action Agency Effect Determination	NMFS Effect Determination
Fin whale	E	35 FR 12222/ December 2, 1970	August 2010	NLAA	NE
Sei whale	E	35 FR 12222/ December 2, 1970	December 2011	NLAA	NE
Sperm whale	E	35 FR 12222/ December 2, 1970	December 2010	NLAA	LAA
Rice's whale	E	84 FR 15446/ April 15, 2019	2020	NLAA	NLAA

We believe the proposed projects will have no effect on the Gulf Sturgeon, smalltooth sawfish, Nassau Grouper, North Atlantic right whale, fin whale, and sei whale, due to each species' recorded geographic ranges and very specific life histories that are not supported in the action area. None of these species are expected to be present in the areas where project effects may occur.

In Section 3.1 below, we discuss potential routes of effect not likely to adversely affect ESA-listed species. In Section 5 below, we discuss potential routes of effect likely to adversely affect ESA-listed species.

Table 6 provides the effects determinations for critical habitat occurring within the action area that the USCG, MARAD, and NMFS believe may be affected by the proposed actions.

Table 6. Effects Determination(s) for Designated Critical Habitat the Action Agencies and NMFS Believe May Be Affected by the Proposed Actions

Species	Critical Habitat in the Action Area	Critical Habitat Rule/Date	Action Agency Effect Determination	NMFS Effect Determination
Sea Turtles				
Loggerhead sea turtle (Northwest Atlantic DPS)	LOGG-S-02 *Sargassum*	79 FR 39856/ July 10, 2014	NE	LAA

In Section 3.6 below, we discuss potential routes of effect not likely to adversely affect designated critical habitat.

In Section 5 below, we discuss potential routes of effect likely to adversely affect designated critical habitat.

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

3.1.1 ESA-Listed Sea Turtles, Whales and Fish

ESA-listed whales are not expected to occur in waters less that 100 m deep in the GOM. The closest area of water depths of 100 m or more are approximately 40 miles offshore from the proposed DWP sites. Therefore, any project-related activities that are confined to the areas in, around, or inshore of the proposed DWPs, including the movement of vessels and equipment, and noise associated with those activities, will have no effect on ESA-listed whales.

ESA-listed sea turtles and fish could be injured or killed if struck by pipeline materials or construction equipment such as the jetting sled proposed for use in pipeline burial for both the SPOT DWP and TGL DWP. We believe this potential effect is extremely unlikely to occur because these species are highly mobile and are expected to move away from any heavy equipment operating in the water. Furthermore, the applicant's implementation of NMFS's Protected Species Construction Conditions (2021) will reduce the risk such adverse effects by requiring all construction workers to watch for ESA-listed species. Operation of any in-water moving equipment will cease immediately if a protected species is seen within a 150-ft radius of the equipment. Activities will not resume until the protected species has departed the project area of its own volition.

ESA-listed sea turtles and fish could be injured or killed if struck by construction and decommissioning related vessels or barges. Due to the species' mobility and the slow speeds at which vessels are required to operate in accordance with the Protected Species Construction Conditions and Vessel Strike Avoidance Measures, the possibility of injury due to a vessel strike from vessels operated by the applicants is extremely unlikely to occur.

Use of the HDD method to tunnel underneath sensitive habitats used by ESA-listed sea turtles and fish will likely avoid direct impacts on these habitats, but could result in indirect impacts to these habitats through an inadvertent release of drilling fluids that could affect water quality within a waterbody. An inadvertent release or "frac-out" occurs when the drilling fluid (composed mostly of water and bentonite clay) finds pathways through natural fissures in the soil and rock along the drill path. Frac-outs that occur beneath water bodies can result in temporary increases in turbidity. SPOT and TGL will implement measures outlined in their HDD Contingency Plans to prevent and minimize such releases.

With these measures in place the likelihood of a frac-out is extremely low, and if a frac-out were to occur, any turbidity effects are likely to be spatially limited and brief in duration. Therefore the potential for ESA-listed sea turtles and fish to be adversely affected by an inadvertent frac-out during construction is extremely unlikely to occur.

The jetting process proposed for pipeline trenching and burial in the Gulf of Mexico by SPOT and TGL is expected to propel sediments approximately 12 ft into the water column above the seafloor. Modeling predicts that offshore pipeline installation would cause sediment deposition greater than 1 millimeter (mm) up to about 656 feet from the trench, with sediment depths ranging from 0.1 mm to greater than 50 mm for the burial of each pipeline. The resulting sediment deposition of greater than 1 mm for the burial of all pipelines would occur over a maximum area of about 6,270 acres for each project. Pile installation and other construction activities at the proposed DWPs may also result in sediment suspension and resettlement, though on a smaller scale than that for pipeline trenching. For example, the model predicted that driving of 72-inch platform piles would result in sediment deposition greater than 1 mm over a maximum area of about 0.02 acre. These disturbances could increase the levels of stress, injury, and mortality of benthic marine species such as isopods, amphipods, mollusks and crabs that may serve as food items for ESA-listed sea turtles and fish. However, due to the dynamic nature of the Gulf of Mexico (i.e., currents, wave and tidal action, and severe storms), these benthic-dwelling organisms are acclimated to high turbidity levels and frequent disturbance, and populations would not be significantly affected by the disturbance caused by the installation of the pipelines. Additionally, these same natural forces (currents and wave action) are expected to redistribute the disturbed sediments, which will quickly restore the natural equilibrium of the sea floor. Given the resiliency of the benthic ecosystem in the action area and the limited duration of these potential effects, any effects to ESA-listed sea turtles and fish foraging success are likely insignificant.

Turbidity levels would gradually decrease with distance and time as sediments settle out of the water column (as described above). ESA-listed sea turtles and fish may be temporarily unable to use the project areas for forage and shelter habitat due to avoidance of construction activities and the related turbidity and noise resulting from these activities. However, we believe any potential effects would be insignificant considering the projects will be located in open-water surrounded by large expanses of similar, nearby habitats, which would allow ESA-listed sea turtles and fish to continue to forage and conduct other essential behaviors in the surrounding area.

Small ESA-listed sea turtles and fish could become impinged or entrained in hydrostatic test water intakes. We believe the potential for this effect to occur is extremely unlikely due to the proposed placement of screening over these intakes and the proposed low inflow velocities.

Discharges of ballast water into the Gulf of Mexico could introduce pollutants or invasive species to a broad area of the Gulf. Changes in water quality or species composition may have the potential to affect ESA-listed sea turtles, whales and fish. In addition, inadvertent spills and leaks from the VLCCs could also have the potential to affect ESA-listed sea turtles, whales and fish in the Gulf of Mexico. For the reasons described below, we believe that any effects from these discharges and inadvertent spills and leaks will be insignificant.

First, all project-related activities associated with the SPOT DWP and TGL DWP will comply with Federal regulations to control the discharge of operational wastes, such as bilge and ballast waters, trash and debris, and sanitary and domestic waste generated from vessels and platforms associated with the proposed Projects. VLCCs and other crude oil carriers may discharge water associated with ballast and cooling water into the GoM while at the DWPs, but domestic or sanitary water onboard the VLCCs or other crude oil carriers would be held in storage tanks. Vessels that release oil or oily ballast water at the DWPs (including within the anchorage area) will be responsible for all cleanup. In addition, vessels transiting to and from the DWPs will be required to abide by the following regulation:

• 33 CFR § 155.370 (Oily Mixture [Bilge Slops]/Fuel Oil Tank Ballast Water Discharges on Oceangoing Ships of 10,000 Gross Tons and Above and Oceangoing Ships of 400 Gross Tons and Above That Carry Ballast Water in Their Fuel Oil Tanks);

Second, VLCCs and other crude oil carriers will manage ballast water in accordance with International Maritime Organization (IMO) Standards. These standards include having a ship-specific ballast water management plan, carrying a record book, and exchange of water mid-ocean or with an on-board ballast water treatment system (IMO 2019b). Mid-ocean exchange of ballast water can flush coastal organisms from foreign ports and thus reduce the potential for introducing non-native species to the port of call. As oil and gas exports have increased from

GoM ports, there has been a corresponding increase in ballast water discharge volumes in U.S. waters, and these discharge volumes are expected to continue to increase into the future (Holzer et al. 2017). Not all vessels operating in GoM waters currently have a ballast water treatment system, but numbers are increasing (Figure 3.3.7-3; National Ballast Information Clearinghouse 2021), and are expected to continue to increase over the life of the proposed SPOT DWP and TGL DWP projects, which is expected to minimize the potential risk of spreading invasive species.

Moreover, VLCCs will manage up to 1.6 million gallons of ballast water per hour for the duration of the loading period, totaling approximately 38 million gallons per ship while moored at the DWPs. Some VLCCs or other crude oil carriers will continue to exchange water mid-ocean during transit, which will not impact water quality in the GoM. Ballast water will be discharged at a maximum rate of 26,667 gpm. Discharge water would be the same temperature as the ambient water temperature and contain a total suspended solids concentration no higher than 30 parts per million. Sedimentation and turbidity from ballast water will not be substantial.

Finally, all VLCCs received at the DWP terminals will be required to implement a ballast water management system in compliance with the IMO rules discussed above. Implementation of these systems is expected to ensure that any effects of ballast water discharges on ESA-listed sea turtles, whales and fish will be insignificant. In light of the limited number of carriers involved in the DWP projects and the conservation measures described above, we believe any effects from ballast water discharges on ESA-listed sea turtles, whales and fish will be insignificant.

Similarly, we believe that effects from any inadvertent oil spills and leaks will be insignificant based on the required implementation of the Shipboard Oil Pollution Emergency Plan described above. Adherence to these plans will minimize the potential for spills and leaks to occur, and ensure that any spills or leaks that do occur will be controlled and cleaned up before they can adversely affect any ESA-listed sea turtles, whales and fish that may be in the area.

Discharges of cooling water into the Gulf of Mexico may increase ambient water temperatures near the VLCC vessels by up to 10°C, which could affect ESA-listed sea turtles and fish. As described above, one VLCC moored for a single loading event would require 400,000 to 530,000 gallons per hour of cooling water. Cooling water will be discharged at a maximum rate of 8,806 gpm, and will typically be approximately 18°F (10°C) above ambient temperatures. Cooling water discharge modeling indicates that at a distance of 328 feet, the discharge plume would be approximately 0.5°F (0.28ºC) to 0.7°F (.39ºC) above surrounding ambient water temperatures. We believe any effect from increases in water temperature associated with discharges of cooling water will be minor, affecting the waters within 328 ft or less (depending on tide and current conditions) of the vessel, and intermittent (1 vessels per day). Therefore, we consider this effect to be insignificant to ESA-listed sea turtles and fish given the mobility of these species, the small volume of cooling water discharged relative to the total volume of water moving through the DWPs, and the limited temperature difference. Any ESA-listed sea turtles and fish that may swim close enough to the cooling water discharge ports to detect the temperature difference are expected to simply swim out of the warmer plume and continue their normal activities in the large expanse of unaffected areas surrounding the VLCC berths.

The increased vessel traffic in the Gulf of Mexico due to VLCC and other crude oil carriers transiting to and from the proposed DWPs could pose an increased risk of accidental vessel strikes for all of the species listed in Table 5 above. We believe it is extremely unlikely that the expected increase in vessel traffic will result in a measureable increase in vessel strikes of these species for a number of reasons. First, VLCCs and other crude oil carriers are expected to use well-established shipping lanes in the Gulf of Mexico (Figure 15). Second, VLCCs and other crude oil carriers are generally slower and generate more noise than typical large vessels, and would therefore be more readily detected and avoided by these mobile species. VLCCs and other crude oil carriers also create a very large bow wave (due to the bulbous bow nose of the vessels), which is likely to push smaller animals (like sea turtles) up and away from the vessel. Finally, to further reduce the potential for vessel strikes, all crude oil carrier captains associated with the proposed projects will be provided, and requested to comply with the NMFS-issued *Vessel Strike Avoidance Measures* (revised February 2021), which includes collision avoidance measures.

The number of vessel transits related to the projects' operations is estimated to be a total of 1,138 vessel transits per year (365 vessels loaded per year for the SPOT DWP + 204 vessels loaded per year for the TGL DWP; each vessel loaded = 2 transits, 1 transit approaching the ports, 1 transit departing the ports). Large vessel traffic throughout the Gulf of Mexico is 964,316 transits per year. When the number of annual vessel transits related to the projects' operations (1,138) is compared with the annual large vessel traffic throughout the Gulf of Mexico (964,316), the proposed actions would result in a very small increase in vessel traffic (1,138 new transits ÷ 964,316 total transits = 0.00118 = 0.118% increase) in the action area.

Sperm whales are by far the most abundant whale occurring in the Gulf of Mexico, and are the only whale with a measurable injury rate due to vessel strikes in the action area. Based on data compiled from the International Whaling Commission Ship Strike Database[1], and supplemented with data from Carrillo and Ritter (2010)[2], NMFS estimates that there is an average of 2 sperm whale strikes per year throughout the entire Gulf of Mexico, with total annual ship transits of approximately 964,316 trips. Given that the proposed operations are expected to result in approximately 1,138 annual transits, this would result in an average of 0.00236 sperm whale strikes per year, or 1 sperm whale strike every 424 years.

X = new sperm whale strikes per year

X ÷ 1,138 = 2 ÷ 964,316

(964,316)X = 2,276

X = 0.00236

1 new strike ÷ 0.00236 strikes per year = 424 years

Given that the entire life of the proposed facilities is estimated at 30 years, we believe the potential for the proposed actions to result in a sperm whale strike in the Gulf of Mexico is extremely unlikely.

Expanding this analysis to all ESA-listed whales throughout all oceans worldwide, a maximum of 1,138 trips per year (if all oil carriers were to travel outside the Gulf) may result from the

[1] http://iwc.int/index.php?cID=872&cType=document

[2] Carrillo, M., and F. Ritter. 2010. Increasing numbers of ship strikes in the Canary Islands: proposals for immediate action to reduce risk of vessel-whale collisions. Journal of Cetacean Research and Management. 11(2): 131–138, 2010

proposed actions. Given that the total number of ships traversing all oceans is much larger than that which occurs within the Gulf, we similarly conclude that the potential for the proposed actions to cause an increase in whale strikes on the open ocean is extremely unlikely.

Regarding Rice's whales in the Gulf of Mexico, there are only a few documented instances of Rice’s whales being struck by vessels in the Gulf since 2006 (Rosel et al. 2016) and (Rosel et al. 2021). However, many Rice’s whale mortalities likely go undetected. Williams et al. (2011) estimate approximately .4% cetacean carcasses are detected in the Gulf of Mexico, when pooled across species.

The primary shipping routes to be followed by the vessels utilizing the proposed DWPs (Figure 15) do not overlap with the core distribution area, where Rice’s whales have been consistently located in the northeastern Gulf of Mexico along the continental shelf between roughly 100 m and 400 m depth. While there is the potential for Rice’s whales to occur in the western Gulf, offshore from the proposed DWPs, in water depths ranging from 100-400 m, (based on passive acoustic monitoring data [Soldevilla et al. (2022), their occurrence outside of the known core distribution area appears to be quite rare. Recent models based on sightings and sightings coupled with passive acoustic monitoring, predict suitable Rice’s whale habitat throughout the Gulf of Mexico, including outside of their core distribution area in the northeastern Gulf of Mexico, generally within the depths of 100 m to 400 m (Farmer et al. 2022), Figure 16). Given that the increased vessel traffic is not expected to traverse through the the species’ core distribution area, the likelihood of a project-related vessel strike of a Rice's whale is considerably lower than that estimated for sperm whales above. Based on the foregoing and the the fact that all crude oil carrier captains associated with the proposed projects will be provided with, and requested to comply with the NMFS-issued *Vessel Strike Avoidance Measures* (revised February 2021), which includes collision avoidance measures to reduce the potential for vessel strikes with ESA-listed species, we believe the risk of a project-related vessel strike involving Rice’s whale is unlikely to occur, and therefore discountable.

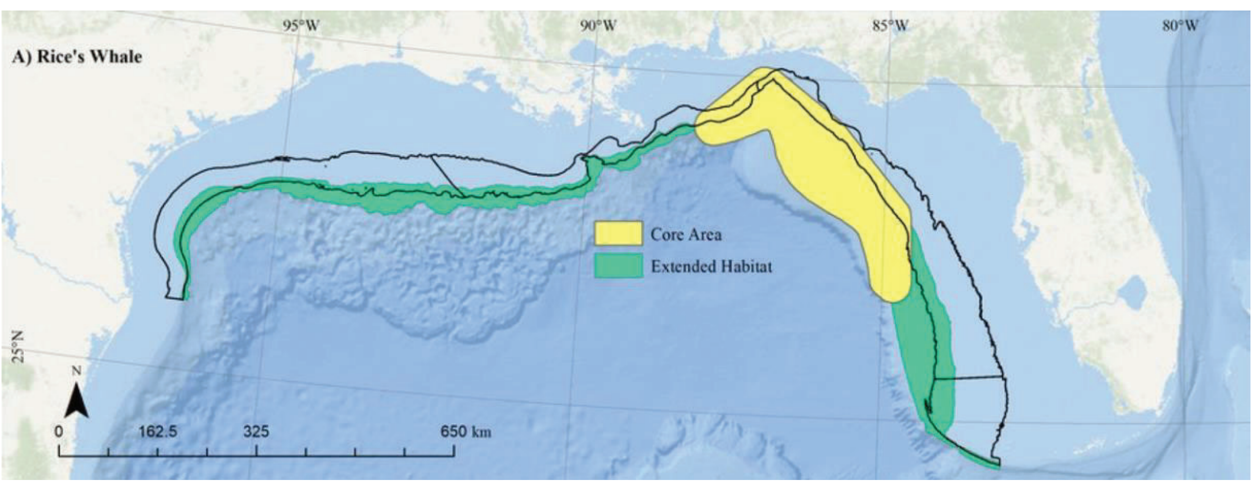


Figure 16. Distribution of Rice's Whale based on sightings (yellow "core area") and sightings coupled with passive acoustic monitoring (green "extended habitat). Adapted from Farmer et al. (2022).

Reliable estimates of overall sea turtle, oceanic whitetip shark, and giant manta strikes throughout the Gulf are not available, but the fact that the proposed action is estimated to result in an increase of just 0.118% of overall shipping transits throughout the Gulf, and an even smaller percentage worldwide, makes the potential for the proposed action to result in an increase in ship strikes on these species extremely unlikely to occur.

In addition to the risk of a vessel strike (addressed above), engine noise from vessels associated with the proposed actions produce acoustic diturbances that may affect ESA-listed species. Given the relatively low magnitude of vessel traffic associated with the proposed operations (approximately 2 to 3 ship per day, traveling along well established shipping lanes), it is extremely unlikely that project vessel opperations would result in extended, chronic exposure to significant noise levels for any individual animals. While longterm, chronic exposure to vessel noise may result in stress and masking of important biological sounds (Clark et al. 2009b; Hatch et al. 2012; Rolland et al. 2012), the passing of 2 or 3 ships per day through a very small portion of these species total range, would not cause significant disturbances, and is not likely to adversely affect ESA-listed whales, sea turtles or fish species.

Pile Driving Noise Analyses
Noise created by pile driving activities can physically injure animals or change animal behavior in the affected areas. Animals can can be physically injured in 2 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. Vibratory pile driving produces continuous, non-pulsed sounds that can be tonal or broadband. 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 webpages 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

NMFS uses the U.S. Navy Phase III criteria for all noise thresholds (U.S. Department of the Navy, 2017). Based on these criteria, potential effects to ESA-listed species may occur when impact or vibratory pile driving produces sounds that exceed the thresholds listed in NMFS Pile Driving Sound Measurement Thresholds for ESA-listed Species in the Southeast Region (https://media.fisheries.noaa.gov/2022-05/2022-05-04_NMFS-Accepted%20Sound%20Measurement%20Thresholds%20Table.pdf). PK and RMS sound pressure are referenced to dB re: 1 μPA, the relative unit used to specify the intensity of sound underwater. Further, SEL and SELcum are referenced to dB re: 1 μPA^2-second. For underwater sounds, a reference pressure of 1 micropascal (μPa) is commonly used to describe sounds in terms of decibels (dB). Thus, 0 dB on the decibel scale would be a measure of sound pressure of 1 µPA.

We use the NMFS Multi-species Pile Driving Tool (https://media.fisheries.noaa.gov/2022-05/BLANK%20Multi-Species%20%28May%202022%29%20external%20%28508%29%20PUBLIC_for%20web.xls) to estimate the radii of physical injury and behavioral effects on ESA-listed species that may be located in the action area based on NMFS-accepted pile driving sound measurement thresholds for species in the NMFS Southeast Region reference above. SPOT proposes to install up to one 72-in diameter pile and up to three 30-in diameter piles via impact hammer per day using bubble curtains as noise abatement. The 72-in diameter piles will require approximately 10,224 strikes per pile, and the 30-in diameter piles will require approximately 408 strikes per pile (see Table 1 above). TGL proposes to install up to one 66-in diameter pile and up to three 54-in diameter piles via impact hammer per day using bubble curtains as noise abatement for the 66-in piles only. In addition, TGL will install up to eight 24-in diameter steel piles via vibratory hammer. The 66-in diameter piles will require approximately 1,261 strikes per pile and the 54-in diameter piles will require approximately 254 strikes per pile (see Table 3 above).

The noise analyses in this consultation evaluates effects to ESA-listed fish and sea turtles identified by NMFS that may be affected by the proposed action. ESA-listed whales are not expected to occur in the shallow water depths where these noise impacts would occur (<150 ft). 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. Because multiple pile-types (i.e., sizes vary from 24-in diameter steel piles to 72-in diameter steel piles) and installation methods (i.e., impact hammer with and without noise abatement measures and vibratory hammer) are proposed, the noise analysis in this consultation evaluates the pile-types and installation methods with the greatest potential effects and largest potential effect radius (i.e., 72-in diameter steel piles via impact hammer with a confined bubble curtain for the SPOT DWP and 54-in diameter steel piles by impact hammer using no bubble curtains for the TGL DWP). Any potential effects of pile driving noise from other proposed pile types and methods would not exceed those described below. Therefore, the potential pile driving noise effects from the other proposed pile types and methods, if any, are expected to occur within a radius of that size or smaller and would result in, at most, the potential effects described below.

For the SPOT DWP, the largest sound impact is produced from the installation of the 72-in diameter platform support steel piles via impact hammer using a confined bubble curtain as a noise abatement measure. Each platform support pile will require 10,224 strikes over a 24-hour driving period. Installation of all 8 of the 72-in diameter piles would take up to 10 days. In order to calculate the approximate impact of the sound disturbances for the 72-in pile installations, the sound measurement values for a similar project reported in Caltrans 2020 were utilized. In the cited proxy project a D-180 Diesel impact hammer was used to install 72-in steel pipe piles in 9 ft (3 m) of water. At a distance of 10 m, the project measured a peak sound pressure level (Peak) of 210 dB, 183 dB sound exposure level (SEL), and 195 dB Root Mean Square (RMS). We afford the platform support pile installation a 10 dB noise reduction due to abatement from the confined bubble curtain system to be utilized during pile installation. This results in final calculated acoustic values used for noise impact calculation of the 72-in piles from a distance of 10 m from the source as 200 dB Peak, 173 dB SEL, and 185 dB RMS.

Based on our noise calculations, installation of 72-in steel piles by impact hammer using a confined bubble system as a noise abatement measure will cause peak pressure injurious noise effects to sea turtles at 0.3 ft (0.1 m) from the pile being driven. Additionally, installation of 72-in steel piles may cause single-strike peak-pressure injurious noise effects up to a distance of 14.4 ft (4.4 m) from the pile for fish. The cumulative sound exposure level of multiple pile strikes over a 24-hour period may cause injury to sea turtles up to 146 ft (44.5 m), and ESA-listed fish up to 1,232.3 ft (375.6 m) away from the pile. To minimize the potential impacts to ESA-listed sea turtles and fish, SPOT will implement a "soft start" procedure to pile driving, which involves ramping up the intensity of the hammer strikes prior to operating at full capacity, and a "shut down" of pile driving activity if an ESA-listed species is observed approaching or within the area of acoustic effects that can produce injury to ESA-listed species.

Additionally, during pile driving activities an approved Protected Species Observer(s) (PSO) will be stationed onboard the pile driving vessel and conduct monitoring for ESA-listed species. NMFS-approved PSOs will be stationed at the site of pile driving activity to monitor for protected species within a pre-determined zone of influence (ZOI). Prior to the start of in-water activities, the PSO will monitor the ZOI for 30 minutes to ensure that the area is clear of all ESA-listed species. The activity would only commence once the PSO has declared that the ZOI is clear. The PSO will monitor the ZOI for the entirety of the in-water activity and record any sightings of protected species. Monitoring will be conducted during daylight and nighttime hours to account for 24-hr pile driving activity. The PSO will use high-quality binoculars during daylight hours, and thermal imaging cameras or night vision binoculars during periods of low or no light situations (nighttime pile driving) to monitor the ZOI. Following the in-water activity, the PSO will conduct post-monitoring of the ZOI for 30 additional minutes. These measures are intended to protect ESA-listed species from potential noise impact related injury if they were to approach the pile installation area. As a result, we believe peak pressure and SELcum injurious noise effects to ESA-listed sea turtles and fish are extremely unlikely to occur due to the mobility of these species. That is, we expect the species to move away from the noise disturbances before the exposure to the noise causes physical injury. Movement away from the injurious sound radius is a behavioral response and is discussed below.

The installation of 72-in steel piles using an impact hammer and a confined bubble curtain as a noise abatement measure could also result in behavioral effects at radii of 7,775.3 ft (2,369.9 m) for ESA-listed fish and 167.5 ft (51.1 m) for sea turtles. Due to the mobility of these species and the open-water environment, we expect the animals to move away from noise disturbances. Because there is similar habitat nearby, we believe behavioral effects will be insignificant. If an animal chooses to remain within the behavioral response zone, it could be exposed to behavioral noise impacts during pile installation. Pile installation activities will be completed in approximately 10 days and ESA-listed species will be able to resume normal activities during quiet periods between pile installations and immediately after completion of the noise producing activities. Therefore, we anticipate that any SPOT DWP related behavioral effects to ESA-listed sea turtles and fish will be insignificant.

For the TGL DWP, using NMFS Multi-species Pile Driving Tool (2021), we determined that the largest impact area will be caused by the installation of the 54-in steel piles by impact hammer with no bubble curtains. Each 54-in pile will require 254 strikes, resulting in 3 piles being driven with approximately 762 strikes per day. Installation of all 18 piles would take up to 6 days. In order to calculate the approximate impact of the sound disturbances for the 54-in pile installations, the sound measurement values for a similar project reported in Caltrans 2020 were utilized. There are no examples of 54-in steel piles in the NMFS Multi-species Pile Driving Tool, so we used the reported sound measurements from the next larger size pile, which were 60-in steel pipe piles, driven in 5 ft (1.5 m) of water. At a distance of 10 m, the peak sound pressure level was 207 dB, 182 dB sound exposure level (SEL), and 192 dB Root Mean Square (RMS).

Based on our noise calculations, installation of 54-in steel piles by impact hammer will cause peak pressure injurious noise effects to sea turtles at 0.7 ft (0.2 m) from the pile being driven. Additionally, installation of 54-in steel piles may cause single-strike peak-pressure injurious noise effects up to a distance of 38.3 ft (11.7 m) from the pile for fish. The cumulative sound exposure level of multiple pile strikes over a 24-hour period may cause injury to sea turtles up to 93.5 ft (28.5 m), and ESA-listed fish up to 1,270.4 ft (387.2 m) away from the pile. To minimize the potential impacts to ESA-listed sea turtles and fish, TGL will implement a “soft start” procedure to pile driving, which involves ramping up the intensity of the hammer strikes prior to operating at full capacity, and a “shut down” of pile driving activity if an ESA-listed species is observed within 150 ft of the pile being driven. Additionally, during pile driving activities an approved Protected Species Observer(s) (PSO) will be stationed onboard the pile driving vessel and conduct monitoring for ESA-listed species. The PSO will monitor the area for the entirety of the in-water activity and record any sightings of protected species. These measures are intended to protect ESA-listed species from potential noise impact related injury if they were to approach the pile installation area. As a result, we believe peak pressure and SELcum injurious noise effects to ESA-listed sea turtles and fish are extremely unlikely to occur due to the mobility of these species. That is, we expect the species to move away from the noise disturbances before the exposure to the noise causes physical injury. Movement away from the injurious sound radius is a behavioral response and is discussed below.

The installation of 54-in steel piles using an impact hammer could also result in behavioral effects at radii 20,700.7 ft (6,309.6 m) for ESA-listed fish and 446 ft (135.9 m) for sea turtles. Due to the mobility of these species and the open-water environment, we expect the animals to move away from noise disturbances. Because there is similar habitat nearby, we believe behavioral effects will be insignificant. If an animal chooses to remain within the behavioral response zone, it could be exposed to behavioral noise impacts during pile installation. Pile installation activities will be completed in approximately 6 days and ESA-listed species will be able to resume normal activities during quiet periods between pile installations and immediately after completion of the noise producing activities. Therefore, we anticipate that any TGL DWP related behavioral effects to ESA-listed sea turtles and fish will be insignificant.

There are several elements of the proposed pile driving operations for both proposed actions that are somewhat novel. NMFS is not aware of any examples of acoustic monitoring data for projects that combine the extreme water depths (over 100 ft), open ocean currents, and the specific bubble curtain designs that will influence the acoustic impacts produced by the proposed projects. We have used the best available information to estimate the expected acoustic impacts of the proposed pile driving operations in concluding that such effects are not reasonably certain to occur, though there remains a level of uncertainty as to the actual level of sound propagation and acoustic impacts that may be experienced by ESA-listed species that may be present in the project areas.

Effects of Oil Spill on Rice's whales
As discussed previously, Rice's whales are primarily concentrated in the northeastern Gulf of Mexico, around DeSoto Canyon (Figure 16). Oil spill modeling discussed below in Section 5 shows that any spills from the DWPs would be expected to drift to the west of the project sites, and none of the modeling indicates the possibility of oil reaching the Rice's whale core distribution area in the northeastern Gulf.

Recent passive acoustic monitoring efforts (Soldevilla et al. (2022) have detected Rice's whale vocalizations in the western Gulf, including the area offshore from the proposed DWPs, where oil spill modeling indicates an extremely low chance of oiling from a spill at either of the DWPs. Given the rarity of detections of Rice's whales in the western Gulf, coupled with the extremely low probability of a large oil spill that could reach out to the areas where Rice's whales might occur (100-400 m depth zone), we believe the potential for a Rice's whale to be adversely affected by an oil spill from the DWPs is extremely unlikely to occur, and therefore discountable.

3.1.2 ESA-listed Corals

Four coral species listed under the ESA occur in the action area: boulder star coral (*Orbicella franksi*), elkhorn coral (*Acropora palmata*), lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolta*). These species occur in the Flower Garden Bank National Marine Sanctuary (FGBNMS) approximately 90 miles to the southeast of the proposed SPOT DWP and TGL DWP.

The abundance of corals in the FGBNMS has been monitored since 1989. FGBNMS has some of the highest percent coral cover in the United States, and unlike other areas, coral cover still dominates benthic communities. In 2016, mean coral cover based on random transects was 49.92 percent within the East Flower Garden Bank (EFGB) study site and 58.54 percent within the West Flower Garden Bank (WFGB) study site (Johnston et al. 2017). Boulder star coral was the principal component of mean percent coral cover within the EFGB study site (20.38 percent) and the WFGB study site (29.29 percent). When Johnston et al. (2017) combined the *Orbicella* species complex, it made up 50.99 percent of the observed coral species within EFGB study sites and 61.67 percent of the observed coral species within WFGB study sites. Boulder star coral covered the greatest total area (58,615,875 cm^3) within EFGB study site surveys and mountainous star coral covered the greatest total area within WFGB (36,290,058 cm^3) study site surveys (Johnston et al. 2017).

We believe it is highly unlikely that the types and amount of marine debris that may originate from the proposed DWPs will ultimately reach and impact corals. Plastic and wood materials will generally float on the surface while any tools or heavier objects will sink directly below the location where they are lost. Because the proposed actions will be located nearly 90 miles from ESA-listed corals on FGBNMS, it is extremely unlikely that any marine debris from either the SPOT DWP or TGL DWP will reach, settle on, or otherwise impact ESA-listed corals. Therefore, we find marine debris from the proposed actions is not likely to adversely affect ESA-listed corals.

Corals are benthic species, and less susceptible to oiling than animals that utilize the water column and surface for feeding, breathing, and swimming. The closest ESA-listed coral species occur approximately 90 miles from the DWPs, within the FGBNMS. Due to the depth of corals in the Flower Gardens Banks (from 55 ft to about 160 ft), and the distance from a potential spill location, we believe the potential for ESA-listed coral to be impacted by oil spilled at or near the DWPs is extremely unlikely to occur. Therefore, we find that potential oil spills resulting from the proposed actions are not likely to adversely affect ESA-listed corals.

3.2 Status of Species Likely to be Adversely Affected

Overview of Status of Sea Turtles

There are 5 species of sea turtles (green [North Atlantic (NA) and South Atlantic (SA) DPSs], hawksbill, Kemp's ridley, leatherback, and loggerhead [Northwest Atlantic (NWA) DPS]) that travel widely throughout the South Atlantic, Gulf of Mexico and the Caribbean. These species are highly migratory and therefore could occur within the action area. Section 3.2.1 will address the general threats that confront all sea turtle species. The remainder of Section 3.2 (Sections 3.2.2 – 3.2.6) will address information on the distribution, life history, population structure, abundance, population trends, and unique threats to each species of sea turtle.

3.2.1 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 sections where appropriate.

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

largest fishery threat to benthic sea turtles in the southeastern United States, and continue to interact with and kill large numbers of sea turtles each year.

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

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

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

Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., 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 Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (DWH Trustees 2015a). 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, or both. The spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. Information on the spill impacts to individual sea turtle species is presented in the Status of the Species sections for each species.

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

Climate Change

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

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

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 2007f). 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. 2006a; Daniels et al. 1993; Fish et al. 2005). The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006a).

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

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

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

3.2.2 **Green Sea Turtle** (Information Relevant to All DPSs)

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 distinct population segments (DPSs) (81 FR 20057 2016) (Figure 17). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific DPSs were listed as threatened. For the purposes of this consultation, only the South Atlantic DPS (SA DPS) and North Atlantic DPS (NA DPS) will be considered, as they are the only two DPSs with individuals occurring in the Atlantic and Gulf of Mexico waters of the United States.

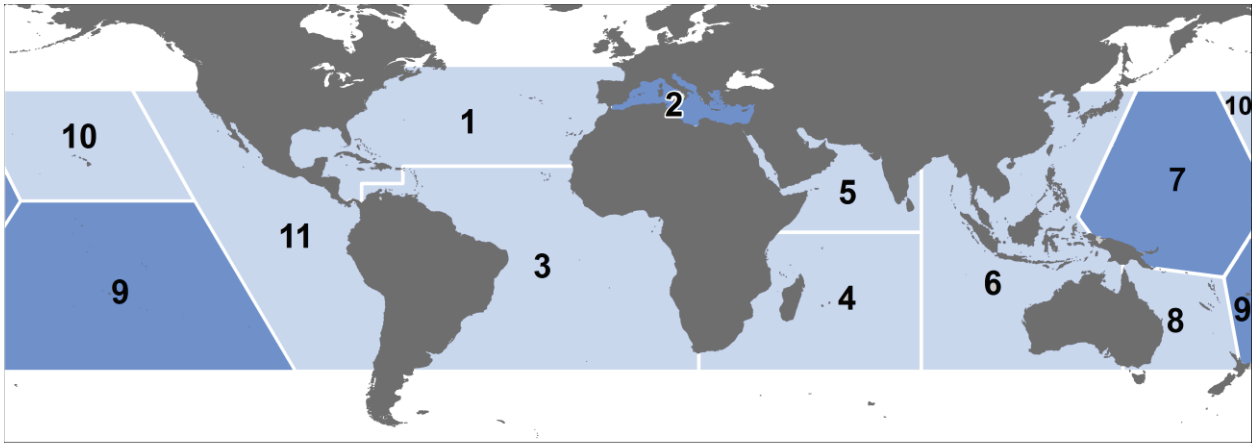


Figure 17. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific

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

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

Differences in mitochondrial DNA properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; FitzSimmons et al. 2006). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Within U.S. waters individuals from both the NA and SA DPSs can be found on foraging grounds. While there are currently no in-depth studies available to determine the percent of NA and SA DPS individuals in any given location, two small-scale studies provide an insight into the degree of mixing on the foraging grounds. An analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of Mexico) found approximately 4% of individuals came from nesting stocks in the SA DPS (specifically Suriname, Aves Island, Brazil, Ascension Island, and Guinea Bissau) (Foley et al. 2007). On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the Aves Island/Suriname nesting assemblage, which is part of the SA DPS (Bass and Witzell 2000). All of the individuals in both studies were benthic juveniles. Available information on green turtle migratory behavior indicates that long distance dispersal is only seen for juvenile turtles. This suggests that larger adult-sized turtles return to forage within the region of their natal rookeries, thereby limiting the potential for gene flow across larger scales (Monzón-Argüello et al. 2010). While all of the mainland U.S. nesting individuals are part of the NA DPS, the U.S. Caribbean nesting assemblages are split between the NA and SA DPS. Nesters in Puerto Rico are part of the NA DPS, while those in the U.S. Virgin Islands are part of the SA DPS. We do not currently have information on what percent of individuals on the U.S. Caribbean foraging grounds come from which DPS.

North Atlantic DPS Distribution

The NA DPS boundary is illustrated in Figure 17. Four regions support nesting concentrations of particular interest in the NA DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

The complete nesting range of NA DPS green sea turtles within the southeastern United States includes sandy beaches between Texas and North Carolina, as well as Puerto Rico (Dow et al. 2007; NMFS and USFWS 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 Mexico waters, green sea turtles are distributed throughout inshore and nearshore waters from Texas to Massachusetts. Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984; Hildebrand 1982; Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1992; Wershoven and Wershoven 1992). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatán Peninsula.

South Atlantic DPS Distribution

The SA DPS boundary is shown in Figure 17, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

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

Life History Information

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

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

While in coastal habitats, green sea turtles exhibit site fidelity to specific foraging and nesting grounds, and it is clear they are capable of "homing in" on these sites if displaced (McMichael et al. 2003). Reproductive migrations of Florida green sea turtles have been identified through flipper tagging and 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 2007b).

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. 2015b), with information for each of the DPSs.

<u>North Atlantic DPS</u>

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

Quintana Roo, Mexico, accounts for approximately 11% of nesting for the DPS (Seminoff et al. 2015b). In the early 1980s, approximately 875 nests/year were deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS 2007g). 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. 2015b). 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 2007b). Modeling by Chaloupka et al. (2008)

using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually.

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

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015b). 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 18). According to data collected from Florida's index nesting beach survey from 1989-2021, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 40,911 in 2019. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011. The pattern departed from the low lows and high peaks in 2020 and 2021 as well, when 2020 nesting only dropped by half from the 2019 high, while 2021 nesting only increased by a small amount over the 2020 nesting (Figure 18).

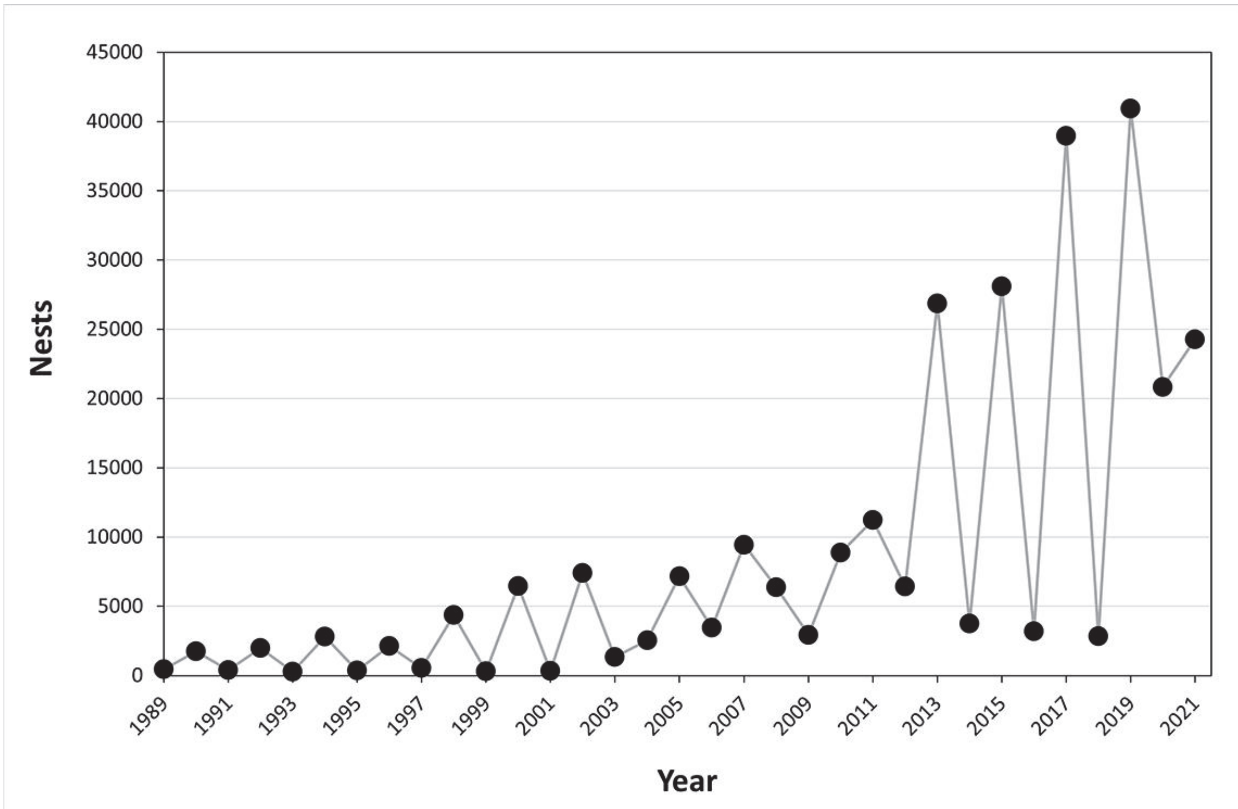


Figure 18. Green sea turtle nesting at Florida index beaches since 1989

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

South Atlantic DPS

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

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

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1.

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

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

Whereas oil spill impacts are discussed generally for all species in Section 3.2.1, specific impacts of the DWH spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juveniles greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (DWH Trustees 2015b). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and dispersants, or both, and loss of foraging resources, which could lead to compromised growth and reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

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

3.2.3 Kemp's Ridley Sea Turtle

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

Species Description and Distribution

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

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

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

Life History Information

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

The average rates of growth may vary by location, but generally fall within 2.2-2.9 ± 2.4 in per year (5.5-7.5 ± 6.2 cm/year) (Schmid and Barichivich 2006; Schmid and Woodhead 2000). Age to sexual maturity ranges greatly from 5-16 years, though NMFS et al. (2011a) 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 19), which indicates the species is recovering.

It is worth noting that when the Bi-National Kemp's Ridley Sea Turtle Population Restoration Project was initiated in 1978, only Rancho Nuevo nests were recorded. In 1988, nesting data from southern beaches at Playa Dos and Barra del Tordo were added. In 1989, data from the northern beaches of Barra Ostionales and Tepehuajes were added, and most recently in 1996, data from La Pesca and Altamira beaches were recorded. Currently, nesting at Rancho Nuevo accounts for just over 81% of all recorded Kemp's ridley nests in Mexico. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley nests in Mexico increased to 21,797 in 2012 (Gladys Porter Zoo 2013). From 2013 through 2014, there was a second significant decline, as only 16,385 and 11,279 nests were recorded, respectively. More recent data, however, indicated an increase in nesting. In 2015 there were 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Gladys Porter Zoo 2016). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm., August 31, 2017), but nesting for 2018 declined to 17,945, with another steep drop to 11,090 nests in 2019 (Gladys Porter Zoo data, 2019). Nesting numbers rebounded in 2020 (18,068 nests) and 2021 (17,671 nests) (CONAMP data, 2021). At this time, it is unclear whether the increases and declines in nesting seen over the past decade represents a population oscillating around an equilibrium point or if nesting will decline or increase in the future.

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

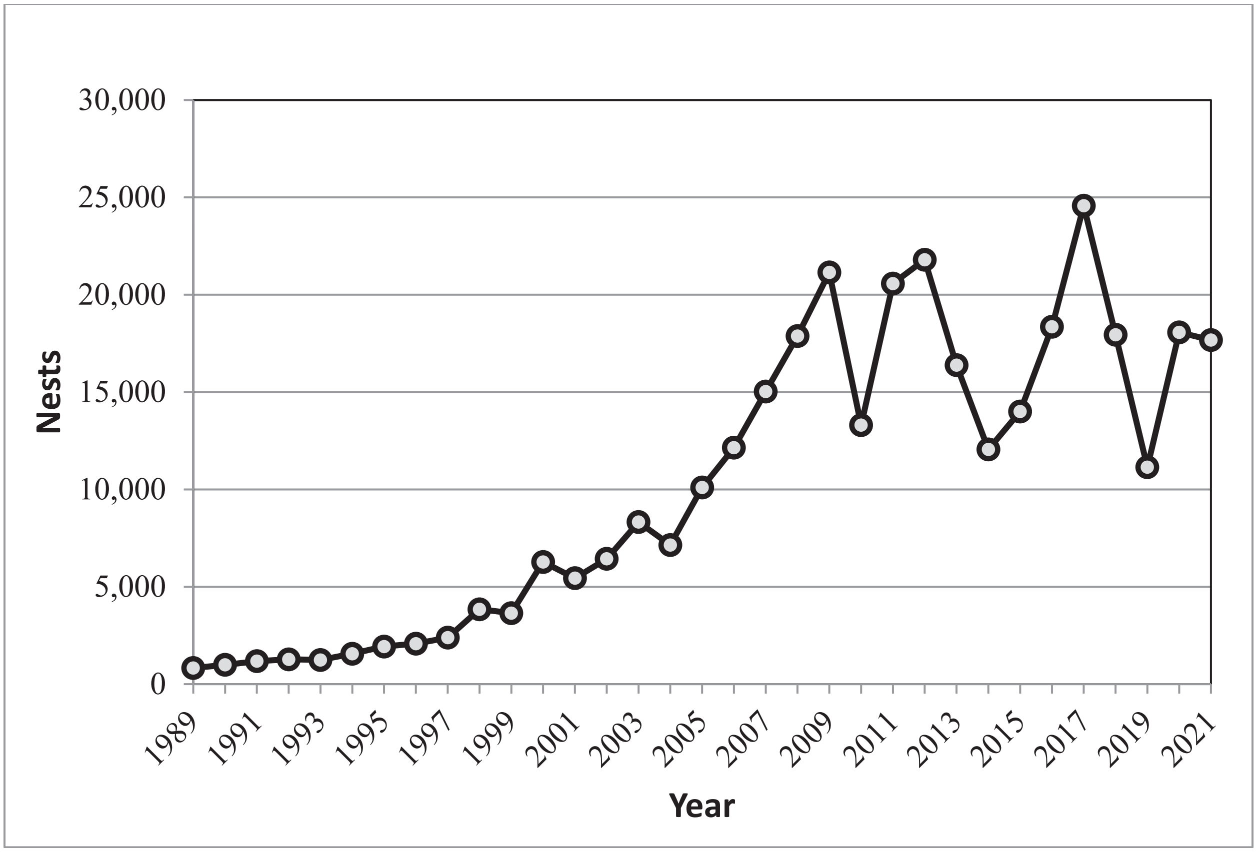


Figure 19. Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019 and CONAMP data 2020, 2021)

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. (2011a) produced an updated model that predicted the population to increase 19% per year and to attain at least 10,000 females nesting on Mexico beaches by 2011. Approximately 25,000 nests would be needed for an estimate of 10,000 nesters on the beach, based on an average 2.5 nests/nesting female. While counts did not reach 25,000 nests by 2015, it is clear that the population has increased over the long term. The increases in Kemp's ridley sea turtle nesting over the last 2 decades is likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of 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 3.2.1; 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.

As Kemp's ridley sea turtles continue to recover and nesting *arribadas*[3] are increasingly established, bacterial and fungal pathogens in nests are also likely to increase. Bacterial and fungal pathogen impacts have been well documented in the large arribadas of the olive ridley at Nancite in Costa Rica (Mo 1988). In some years, and on some sections of the beach, the hatching success can be as low as 5% (Mo 1988). As the Kemp's ridley nest density at Rancho Nuevo and adjacent beaches continues to increase, appropriate monitoring of emergence success will be necessary to determine if there are any density-dependent effects.

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

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

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

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-inch (in) bar spacing for all vessels using skimmer trawls, pusher-head trawls, or wing nets. Ultimately, we published a final rule on December 20, 2019 (84 FR 70048), that requires all skimmer trawl vessels 40 feet and greater in length to use TEDs designed to exclude small sea turtles in their nets effective April 1, 2021. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of Mexico may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 3.2.1, 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. 2011a), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of Mexico throughout their lives (DWH Trustees 2016).

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

3.2.4 Leatherback Sea Turtle

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

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

Unlike other sea turtles, leatherbacks have several unique traits that enable them to live in cold water. For example, leatherbacks have a countercurrent circulatory system (Greer et al. 1973),[4] a thick layer of insulating fat (Davenport et al. 1990; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),[5] and they can increase their body temperature through increased metabolic activity (Bostrom and Jones 2007; Southwood et al. 2005). These adaptations allow leatherbacks to be comfortable in a wide range of temperatures, which helps them to travel further than any

[4] Countercurrent circulation is a highly efficient means of minimizing heat loss through the skin's surface because heat is recycled. For example, a countercurrent circulation system often has an artery containing warm blood from the heart surrounded by a bundle of veins containing cool blood from the body's surface. As the warm blood flows away from the heart, it passes much of its heat to the colder blood returning to the heart via the veins. This conserves heat by recirculating it back to the body's core.
[5] "Gigantothermy" refers to a condition when an animal has relatively high volume compared to its surface area, and as a result, it loses less heat.

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

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

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

Life History Information

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

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

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

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

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

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

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

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

Researchers believe the cyclical pattern of beach erosion and then reformation has affected leatherback nesting patterns in the Guianas. For example, between 1979 and 1986, the number of leatherback nests in French Guiana had increased by about 15% annually (NMFS 2001). This increase was then followed by a nesting decline of about 15% annually. This decline corresponded with the erosion of beaches in French Guiana and increased nesting in Suriname. This pattern suggests that the declines observed since 1987 might actually be a part of a nesting cycle that coincides with cyclic beach erosion in Guiana (Schulz 1975). Researchers think that the cycle of erosion and reformation of beaches may have changed where leatherbacks nest throughout this region. The idea of shifting nesting beach locations was supported by increased nesting in Suriname,[6] while the number of nests was declining at beaches in Guiana (Hilterman et al. 2003). This information suggested the long-term trend for the overall Suriname and French Guiana population was increasing. A more recent cycle of nesting declines from 2008-2017, as

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

high at 31% annual decline in the Awala-Yalimapo area of French Guiana and almost 20% annual declines in Guyana, has changed the long-term nesting trends in the region negative as described above (Northwest Atlantic Leatherback Working Group 2018).

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

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

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. FWC Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 20 and Table 7). A similar pattern was also observed statewide (Table 7). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting. Overall, the trend showed growth on Florida's east coast beaches. Tiwari et al. (2013) report an annual growth rate of 9.7% and a three-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, with a partial rebound in 2018. The annual geometric mean trend for Florida has been a decline of almost 7% from 2008-2017, but the long-term trend (1990-2017) remains positive with an annual geometric mean increase of over 9% (Northwest Atlantic Leatherback Working Group 2018).

Table 7. Number of Leatherback Sea Turtle Nests in Florida

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

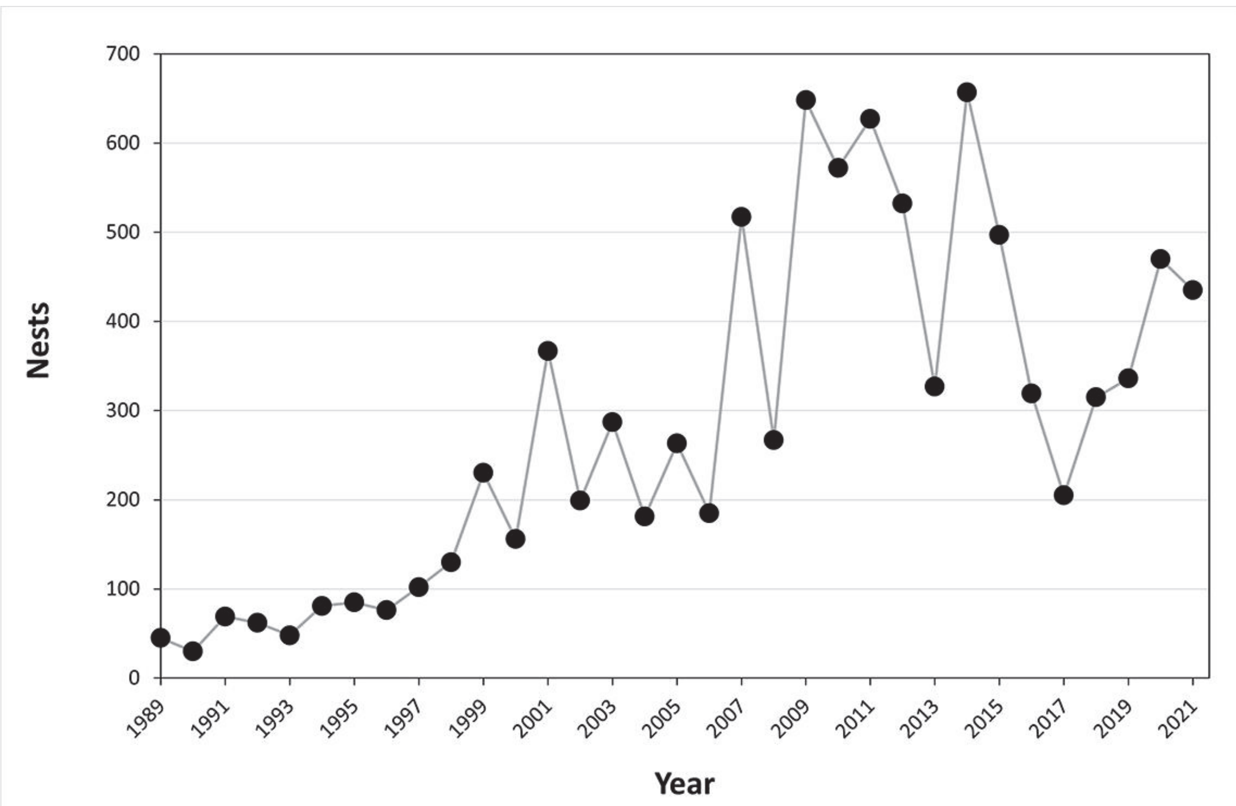


Figure 20. Leatherback sea turtle nesting at Florida index beaches since 1989

The West African nesting stock of leatherbacks is large and important, but it is a mostly unstudied aggregation. Nesting occurs in various countries along Africa's Atlantic coast, but much of the nesting is undocumented and the data are inconsistent. Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in a single season

(Fretey et al. 2007). Fretey et al. (2007) provide detailed information about other known nesting beaches and survey efforts along the Atlantic African coast. Because of the lack of consistent effort and minimal available data, trend analyses were not possible for this stock (TEWG 2007).

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

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

Threats

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

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

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

As discussed in Section 3.2.1, global climate change can be expected to have various impacts on all sea turtles, including leatherbacks. Global climate change is likely to also influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007e). Several studies have shown leatherback distribution is influenced by jellyfish abundance (Houghton et al. 2006; Witt et al. 2007; Witt et al. 2006); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 3.2.1, specific impacts of the DWH oil spill on leatherback sea turtles are considered here. Available information indicates leatherback sea turtles (along with hawksbill turtles) were likely directly affected by the oil spill. Leatherbacks were documented in the spill area, but the number of affected leatherbacks was not estimated due to a lack of information compared to other species. Given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (TEWG 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, it was concluded that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. Potential DWH-related impacts to leatherback sea turtles include direct oiling or contact with dispersants from surface and subsurface oil and dispersants, inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil or dispersants, or both, and loss of foraging resources which could lead to compromised growth and reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts likely occurred to leatherbacks, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event may be relatively low. Thus, a population-level impact may not have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

3.2.5 Loggerhead Sea Turtle – Northwest Atlantic DPS

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

Species Description and Distribution

Loggerheads are large sea turtles. Adults in the southeast United States average about 3 ft (92 cm) long, measured as a straight carapace length (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 Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba (Moncada Gavilan 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

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

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

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

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

Life History Information
The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, which include the ecosystems those stages generally use: (1) egg (terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone[7]), (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 2008a). 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 2008a). Loggerhead hatchlings are 1.5-2 inches long and weigh about 0.7 oz (20 g).

As post-hatchlings, loggerheads hatched on U.S. beaches enter the "oceanic juvenile" life stage, migrating offshore and becoming associated with *Sargassum* habitats, driftlines, and other convergence zones (Carr 1986; Conant et al. 2009b; Witherington 2002). Oceanic juveniles

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

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

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

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. 2009b).

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

Status and Population Dynamics

A number of stock assessments and similar reviews (Conant et al. 2009b; Heppell et al. 2003a; NMFS-SEFSC 2009a; NMFS 2001; NMFS and USFWS 2008a; 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 2008a). NMFS and USFWS (2008a) concluded that the lack of change in 2 important demographic parameters of loggerheads, remigration interval and clutch frequency, indicate that time series on numbers of nests can provide reliable information on trends in the female population.

<u>Peninsular Florida Recovery Unit</u>

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

In addition to the total nest count estimates, the Florida Fish and Wildlife Research Institute (FWRI) uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. FWRI uses the standardized index survey data to analyze the nesting trends (Figure 21) (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 confidence intervals. Nesting at the core index beaches declined in 2017 to 48,033, and rose again each year through 2020, reaching 53,443 nests before dipping back to 49,100 in 2021. It is important to note that with the wide confidence intervals and uncertainty around the variability in nesting parameters (changes and variability in nests/female, nesting intervals, etc.) it is unclear whether the nesting trend equates to an increase in the population or nesting females over that time frame (Ceriani et al. 2019).

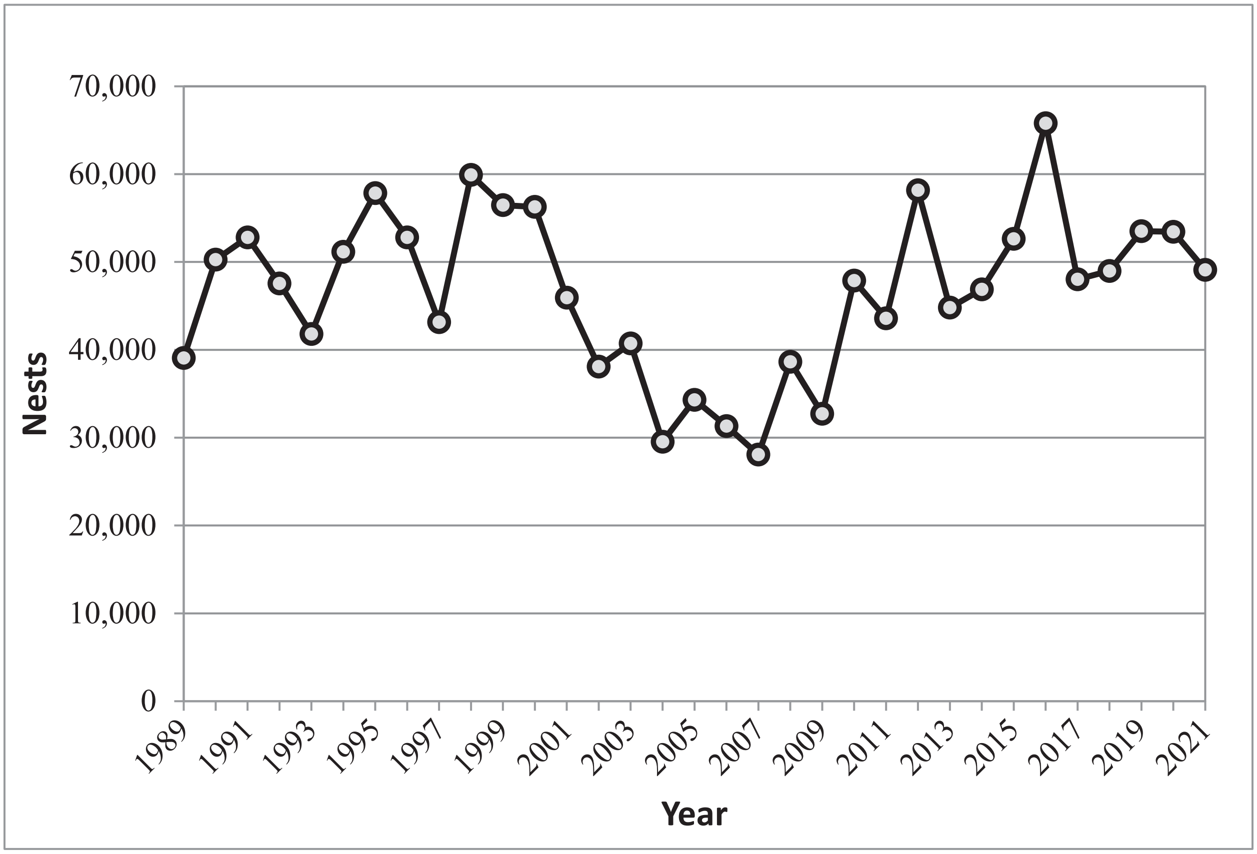

Figure 21. Loggerhead sea turtle nesting at Florida index beaches since 1989

Northern Recovery Unit

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

Data since that analysis (Table 8) 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.

Table 8. Total Number of NRU Loggerhead Nests (GADNR, SCDNR, and NCWRC nesting datasets compiled at Seaturtle.org)

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

South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record. After another drop in 2018, a new record was set for the 2019 season, with a return to 2016 levels in 2020 and 2021 (Figure 22).

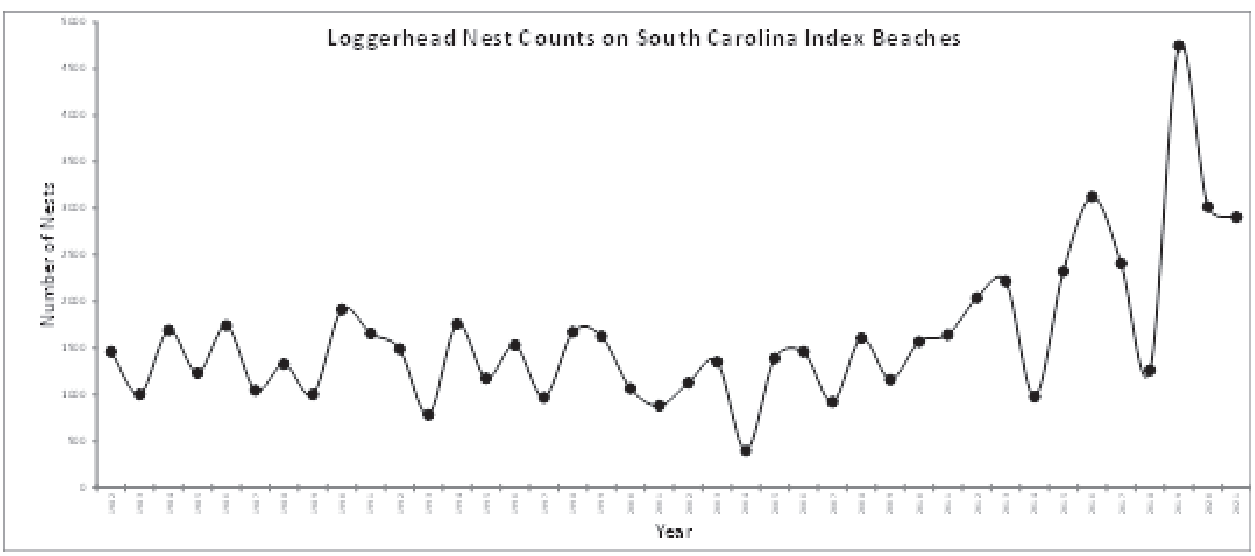

Figure 22. South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website: https://www.dnr.sc.gov/seaturtle/ibs.htm)

Other Northwest Atlantic DPS Recovery Units

The remaining 3 recovery units—Dry Tortugas (DTRU), Northern Gulf of Mexico (NGMRU), and Greater Caribbean (GCRU)—are much smaller nesting assemblages, but they are still considered essential to the continued existence of the species. Nesting surveys for the DTRU are conducted as part of Florida's statewide survey program. Survey effort was relatively stable during the 9-year period from 1995-2004, although the 2002 year was missed. Nest counts ranged from 168-270, with a mean of 246, but there was no detectable trend during this period (NMFS and USFWS 2008a). Nest counts for the NGMRU are focused on index beaches rather than all beaches where nesting occurs. Analysis of the 12-year dataset (1997-2008) of index nesting beaches in the area shows a statistically significant declining trend of 4.7% annually. Nesting on the Florida Panhandle index beaches, which represents the majority of NGMRU nesting, had shown a large increase in 2008, but then declined again in 2009 and 2010 before rising back to a level similar to the 2003-2007 average in 2011. From 1989-2018 the average number of NGMRU nests annually on index beaches was 169 nests, with an average of 1100 counted in the statewide nesting counts (Ceriani et al. 2019). Nesting survey effort has been inconsistent among the GCRU nesting beaches, and no trend can be determined for this subpopulation (NMFS and USFWS 2008a). 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 2008a).

In-water Trends

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

Population Estimate

The NMFS Southeast Fisheries Science Center developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS-SEFSC 2009a). 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 2009a). 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 2009a). 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 3.2.1. 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 NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant et al. 2009b).

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

While oil spill impacts are discussed generally for all species in Section 3.2.1, specific impacts of the Deepwater Horizon (DWH) oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (DWH Trustees 2015b). 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, or both, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

Unlike Kemp's ridleys, the majority of nesting for the NWA DPS occurs on the Atlantic coast and, thus, loggerheads were impacted to a relatively lesser degree. However, it is likely that impacts to the NGMRU of the NWA DPS would be proportionally much greater than the impacts occurring to other recovery units. Impacts to nesting and oiling effects on a large proportion of the NGMRU recovery unit, especially mating and nesting adults likely had an impact on the NGMRU. Based on the response injury evaluations for Florida Panhandle and Alabama nesting beaches (which fall under the NGMRU), the DWH Trustees (2016) estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH oil spill response activities on nesting beaches. Although the long-term effects remain unknown, the DWH oil spill event impacts to the Northern Gulf of Mexico Recovery Unit may result in some nesting declines in the future due to a large reduction of oceanic age classes during the DWH oil spill event. Although adverse impacts occurred to loggerheads, the proportion of the population that is expected to have been exposed to and directly impacted by the DWH oil spill event is relatively low. Thus we do not believe a population-level impact occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

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

3.2.6 Hawksbill Sea Turtle

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

Species Description and Distribution

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

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

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

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

Life History Information

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

Female hawksbills return to the beaches where they were born (natal beaches) every 2-3 years to nest (Van Dam et al. 1991; Witzell 1983) and generally lay 3-5 nests per season (Richardson et al. 1999). Compared with other sea turtles, the number of eggs per nest (clutch) for hawksbills can be quite high. The largest clutches recorded for any sea turtle belong to hawksbills (approximately 250 eggs per nest) ((Hirth and Latif 1980), though nests in the U.S. Caribbean and Florida more typically contain approximately 140 eggs (USFWS hawksbill fact sheet, https://www.fws.gov/species/carey-eretmochelys-imbricata). Eggs incubate for approximately 60 days before hatching (USFWS hawksbill fact sheet). Hatchling hawksbill sea turtles typically measure 1-2 in (2.5-5 cm) in length and weigh approximately 0.5 oz (15 g).

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

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

Status and Population Dynamics

There are currently no reliable estimates of population abundance and trends for non-nesting hawksbills at the time of this consultation; therefore, nesting beach data is currently the primary information source for evaluating trends in global abundance. Most hawksbill populations around the globe are either declining, depleted, and remnants of larger aggregations (NMFS and USFWS 2007c). The largest nesting population of hawksbills occurs in Australia where approximately 2,000 hawksbills nest off the northwest coast and about 6,000-8,000 nest off the Great Barrier Reef each year (Spotila 2004). Additionally, about 2,000 hawksbills nest each year in Indonesia and 1,000 nest in the Republic of Seychelles (Spotila 2004). In the United States, hawksbills typically laid about 500-1,000 nests on Mona Island, Puerto Rico in the past (Diez and Van Dam 2007), but the numbers appear to be increasing, as the Puerto Rico Department of

Natural and Environmental Resources counted nearly 1,600 nests in 2010 (PRDNER nesting data). Another 56-150 nests are typically laid on Buck Island off St. Croix (Meylan 1999b; Mortimer and Donnelly 2008). Nesting also occurs to a lesser extent on beaches on Culebra Island and Vieques Island in Puerto Rico, the mainland of Puerto Rico, and additional beaches on St. Croix, St. John, and St. Thomas, U.S. Virgin Islands.

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

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

Threats

Hawksbills are currently subjected to the same suite of threats on both nesting beaches and in the marine environment that affect other sea turtles (e.g., interaction with federal and state fisheries, coastal construction, oil spills, climate change affecting sex ratios) as discussed in Section 3.2.1. There are also specific threats that are of special emphasis, or are unique, for hawksbill sea turtles discussed in further detail below.

While oil spill impacts are discussed generally for all species in Section 3.2.1, specific impacts of the DWH spill on hawksbill turtles have been estimated. Hawksbills made up 2.2% (8,850) of small juvenile sea turtle (of those that could be identified to species) exposures to oil in offshore areas, with an estimate of 615 to 3,090 individuals dying as a result of the direct exposure (DWH Trustees 2015b). No quantification of large benthic juveniles or adults was made. Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil or dispersants, or both, and loss of foraging resources which could lead to compromised growth and reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts occurred to hawksbills, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event is relatively low, and thus a population-level impact is not believed to have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

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

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

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

3.2.7 Giant manta

NMFS listed the giant manta ray (*Manta birostris*) as threatened under the ESA (83 FR 2916, Publication Date January 22, 2018) and determined that the designation of critical habitat is not prudent on (84 FR 66652, Publication Date December 5, 2019). On December 4, 2019, NMFS published a recovery outline for the giant manta ray (NMFS 2019), which serves as an interim guidance to direct recovery efforts for giant manta ray.

Species Description and Distribution

The giant manta ray is the largest living ray, with a wingspan reaching a width of up to 7 m (23 ft), and an average size between 4-5 m (15-16.5 ft). The giant manta ray is recognized by its large diamond-shaped body with elongated wing-like pectoral fins, ventrally placed gill slits, laterally placed eyes, and wide terminal mouth. In front of the mouth, it has 2 structures called cephalic lobes that extend and help to introduce water into the mouth for feeding activities (making them the only vertebrate animals with 3 paired appendages). Giant manta rays have 2 distinct color types: chevron (mostly black back dorsal side and white ventral side) and black (almost completely black on both ventral and dorsal sides). Most of the chevron variants have a black dorsal surface and a white ventral surface with distinct patterns on the underside that can be used to identify individuals (Miller and Klimovich 2017). There are bright white shoulder markings on the dorsal side that form 2 mirror image right-angle triangles, creating a T-shape on the upper shoulders.

The giant manta ray is found worldwide in tropical and subtropical oceans and in productive coastal areas. They also occasionally occur within estuaries (e.g., lagoons and bays) and Intracostal Waterways (ICWW). In terms of range, within the Northern hemisphere, the species has been documented as far north as southern California and New Jersey on the United States west and east coasts, respectively, and Mutsu Bay, Aomori, Japan, the Sinai Peninsula and Arabian Sea, Egypt, and the Azores Islands (CITES 2013; Gudger 1922; Kashiwagi et al. 2010; Moore 2012). In the Southern Hemisphere, the species occurs as far south as Peru, Uruguay, South Africa, New Zealand and French Polynesia (CITES 2013; Mourier 2012). Within its range, the giant manta ray inhabits tropical, subtropical, and temperate bodies of water and is commonly found offshore, in oceanic waters, and near productive coastlines (Figure 23) (Kashiwagi et al. 2011; Marshall et al. 2009).

Figure 23. The Extent of Occurrence (dark blue) and Area of Occupancy (light blue) based on species distribution (Lawson et al. 2017).

Life History Information
Giant manta rays make seasonal long-distance migrations, aggregate in certain areas and remain resident, or aggregate seasonally (Dewar et al. 2008; Girondot et al. 2015; Graham et al. 2012; Stewart et al. 2016b). The giant manta ray is a seasonal visitor along productive coastlines with regular upwelling, in oceanic island groups, and at offshore pinnacles and seamounts. The timing of these visits varies by region and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior. They have also been observed in estuarine waters inlets, with use of these waters as potential nursery grounds (Adams and Amesbury 1998; Medeiros et al. 2015; Milessi and Oddone 2003).

Giant manta rays are known to aggregate in various locations around the world in groups usually ranging from 100-1,000 (Graham et al. 2012; Notarbartolo di Sciara and Hillyer 1989; Venables 2013). These sites function as feeding sites, cleaning stations, or sites where courtship interactions take place (Graham et al. 2012; Heinrichs et al. 2011; Venables 2013). The appearance of giant manta rays in these locations is generally predictable. For example, food availability due to high productivity events tends to play a significant role in feeding site aggregations (Heinrichs et al. 2011; Notarbartolo di Sciara and Hillyer 1989). Giant manta rays have also been shown to return to a preferred site of feeding or cleaning over extended periods of time (Dewar et al. 2008; Graham et al. 2012; Medeiros et al. 2015). In addition, giant and reef manta rays in Keauhou and Ho”ona Bays in Hawaii, appear to exhibit learned behavior. These manta rays learned to associate artificially lighting with high plankton concertation (primary food source) and shifted foraging strategies to include sites that had artificially lighting at night (Clark 2010). While little is known about giant manta ray aggregation sites, the Flower Garden Banks National Marine Sanctuary and the surrounding region might represent the first documented nursery habitat for giant manta ray (Stewart et al. 2018). Stewart et al. (2018) found that the Flower Garden Banks National Marine Sanctuary provides nursery habitat for juvenile giant manta rays because small age classes have been observed consistently across years at both the population and individual level. The Flower Garden Banks National Marine Sanctuary may be an optimal nursery ground because of its location near the edge of the continental shelf and proximity to abundant pelagic food resources. In addition, small juveniles are frequently observed along a portion of Florida's east coast, indicating that this area may also function as a nursery ground for juvenile giant manta rays. Since directed visual surveys began in 2016, juvenile giant manta rays are regularly observed in the shallow waters (less than 5 m depth) from Jupiter Inlet to Boynton Beach Inlet (J Pate, Florida Manta Project, unpublished data). However, the extent of this purported nursery ground is unknown as the survey area is limited to a relatively narrow geographic area along Florida's southeast coast.

The giant manta ray appears to exhibit a high degree of plasticity in terms of its use of depths within its habitat. Tagging studies have shown that the giant manta rays conduct night descents from 200-450m depths (Rubin et al. 2008; Stewart et al. 2016b) and are capable of diving to depths exceeding 1,000 m (A. Marshall et al. unpublished data 2011, cited in Marshall et al. (2011)). Stewart et al. (2016b) found diving behavior may be influenced by season, and more specifically, shifts in prey location associated with the thermocline, with tagged giant manta rays (n=4) observed spending a greater proportion of time at the surface from April to June and in deeper waters from August to September. Overall, studies indicate that giant manta rays have a more complex depth profile of their foraging habitat than previously thought, and may actually be supplementing their diet with the observed opportunistic feeding in near-surface waters (Burgess et al. 2016; Couturier et al. 2013).

Giant manta rays primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderately sized fishes (Miller and Klimovich 2017). While it was previously assumed, based on field observations, that giant manta rays feed predominantly during the day on surface zooplankton, results from recent studies (Burgess et al. 2016; Couturier et al. 2013) indicate that these feeding events are not an important source of the dietary intake. When feeding, giant manta rays hold their cephalic lobes in an "O" shape and open their mouth wide, which creates a funnel that pushes water and prey through their mouth and over their gill rakers. They use many different types of feeding strategies, such as barrel rolling (doing somersaults repeatedly) and creating feeding chains with other mantas to maximize prey intake.

The giant manta ray is viviparous (i.e., gives birth to live young). They are slow to mature and have very low fecundity and typically give birth to only one pup every 2 to 3 years. Gestation lasts approximately 10-14 months. Females are only able to produce between 5 and 15 pups in a lifetime (CITES 2013; Miller and Klimovich 2017). The giant manta ray has one of the lowest maximum population growth rates of all elasmobranchs (Dulvy et al. 2014; Miller and Klimovich 2017). The giant manta rays generation time (based on *M. alfredi* life history parameters) is estimated to be 25 years (Miller and Klimovich 2017).

Although giant manta rays have been reported to live at least 40 years, not much is known about their growth and development. Maturity is thought to occur between 8-10 years of age (Miller and Klimovich 2017). Males are estimated to mature at around 3.8 m disc width (slightly smaller than females) and females at 4.5 m disc width (Rambahiniarison et al. 2018).

Status and Population Dynamics

There are no current or historical estimates of global abundance of giant manta rays, with most estimates of subpopulations based on anecdotal observations. The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 2013) found that only ten populations of giant manta rays had been actively studied, 25 other aggregations have been anecdotally identified, all other sightings are rare, and the total global population may be small. Subpopulation abundance estimates range between 42 and 1,500 individuals, but are anecdotal and subject to bias (Miller and Klimovich 2017). The largest subpopulations and records of individuals come from the Indo-Pacific and eastern Pacific. Ecuador is thought to be home to the largest identified population (n=1,500) of giant manta rays in the world, with large aggregation sites within the waters of the Machalilla National Park and the Galapagos Marine Reserve (Hearn et al. 2014). Within the Indian Ocean, numbers of giant manta rays identified through citizen science in Thailand's waters (primarily on the west coast, off Khao Lak and Koh Lanta) was 288 in 2016. These numbers reportedly surpass the estimate of identified giant mantas in Mozambique (n=254), possibly indicating that Thailand may be home to the largest aggregation of giant manta rays within the Indian Ocean (MantaMatcher 2016). Miller and Klimovich (2017) concluded that giant manta rays are at risk throughout a significant portion of their range, due in large part to the observed declines in the Indo-Pacific. There have been decreases in landings of up to 95% in the Indo-Pacific, although similar declines have not been observed in areas with other subpopulations, such as Mozambique and Ecuador. In the U.S. Atlantic and Caribbean, giant manta ray sightings are concentrated along the east coast as far north as New Jersey, within the Gulf of Mexico, and off the coasts of the U.S. Virgin Islands and Puerto Rico. Because most sightings of the species have been opportunistic during other surveys, researchers are still unsure what attracts giant manta rays to certain areas and not others and where they go for the remainder of the time (84 FR 66652; Publication Date December 5, 2019).

The available sightings data indicate that giant manta rays occur regularly along Florida's east coast. In 2010, Georgia Aquarium began conducting aerial surveys for giant manta rays. The surveys are conducted in spring and summer and run from the beach parallel to the shoreline (0 to 2.5 nautical miles), from St. Augustine Beach Pier to Flagler Beach Pier, Florida. The numbers, location, and peak timing of the manta rays to this area varies by year (H. Webb unpublished data). In addition, juvenile giant manta rays have also been regularly observed inshore off the southeast Florida. Since 2016, researchers with the Marine Megafauna Foundation have been conducting annual surveys along a small transect off Palm Beach, Florida, between Jupiter Inlet and Boynton Beach Inlet (~44 km, 24 nautical miles) (J. Pate, MMF, pers. comm. to M. Miller, NMFS OPR, 2018). Results from these surveys indicate that juvenile manta rays are present in these waters for the majority of the year (observations span from May to December), with re-sightings data that suggest some manta rays may remain in the area for extended periods of time or return in subsequent years (J. Pate unpublished data). In the Gulf of Mexico, within the Flower Garden Banks National Marine Sanctuary, 95 unique individuals have been recorded between 1982 and 2017 (Stewart et al. 2018).

Threats

The giant manta ray faces many threats, including fisheries interactions, environmental contaminants (microplastics, marine debris, petroleum products, etc.), vessel strikes, entanglement, and global climate change. Overall, the predictable nature of their appearances, combined with slow swimming speed, large size, and lack of fear towards humans, may increase their vulnerability to threats (Convention on Migratory Species 2014; O'Malley et al. 2013). The ESA status review determined that the greatest threat to the species results from fisheries related mortality (Miller and Klimovich 2017); (83 FR 2916, Publication Date January 22, 2018).

Commercial Harvest and Fisheries Bycatch

Commercial harvest and incidental bycatch in fisheries is cited as the primary cause for the decline in the giant manta ray and threat to future recovery (Miller and Klimovich 2017). We anticipate that these threats will continue to affect the rate of recovery of the giant manta ray. Worldwide giant manta ray catches have been recorded in at least 30 large and small-scale fisheries covering 25 countries (Lawson et al. 2016). Demand for the gills of giant manta rays and other mobula rays has risen dramatically in Asian markets. With this expansion of the international gill raker market and increasing demand for manta ray products, estimated harvest of giant manta rays, particularly in many portions of the Indo-Pacific, frequently exceeds numbers of identified individuals in those areas and are accompanied by observed declines in sightings and landings of the species of up to 95% (Miller and Klimovich 2017). In the Indian Ocean, manta rays (primarily giant manta rays) are mainly caught as bycatch in purse seine and gillnet fisheries (Oliver et al. 2015). In the western Indian Ocean, data from the pelagic tuna purse seine fishery suggests that giant manta and mobula rays, together, are an insignificant portion of the bycatch, comprising less than 1% of the total non-tuna bycatch per year (Chassot et al. 2008; Romanov 2002). In the U.S., bycatch of giant manta rays has been recorded in the coastal migratory pelagic gillnet, gulf reef fish bottom longline, Atlantic shark gillnet, pelagic longline, pelagic bottom longline, and trawl fisheries. Incidental capture of giant manta ray is also a rare occurrence in the elasmobranch catch within U.S. Atlantic and Gulf of Mexico, with the majority that are caught released alive. In addition to directed harvest and bycatch in commercial fisheries, the giant manta ray is incidentally captured by recreational fishers using vertical line (i.e., handline, bandit gear, and rod-and-reel). Researchers frequently report giant manta rays having evidence of recreational gear interactions along the east coast of Florida (i.e., manta rays have embedded fishing hooks with attached trailing monofilament line) (J. Pate, Florida Manta Project, unpublished data). Internet searches also document recreational interactions with giant manta rays. For example, recreational fishers will search for giant manta rays while targeting cobia, as cobia often accompany giant manta rays (anglers will cast at manta rays in an effort to hook cobia). In addition, giant manta rays are commonly observed swimming near or underneath public fishing piers where they may become foul-hooked. The current threat of mortality associated with recreational fisheries is expected to be low, given that we have no reports of recreational fishers retaining giant manta ray. However, bycatch in recreational fisheries remains a potential threat to the species.

Vessel Strike

Vessel strikes can injure or kill giant manta rays, decreasing fitness or contributing to non-natural mortality (Couturier et al. 2012; Deakos et al. 2011). Giant manta rays do not surface to breath, but they can spend considerable time in surface waters, while basking and feeding, where they are more susceptible to vessel strikes (McGregor et al., 2019). They show little fear toward vessels which can also make them extremely vulnerable to vessel strikes (Deakos 2010). Five giant manta rays were reported to have been struck by vessels from 2016 through 2018; individuals had injuries (i.e., fresh or healed dorsal surface propeller scars) consistent with a vessel strike. These interactions were observed by researchers conducting surveys from Boynton

Beach to Jupiter, Florida (J. Pate, Florida Manta Project, unpublished data). The giant manta ray is frequently observed in nearshore coastal waters and feeding within and around inlets. As vessel traffic is concentrated in and around inlets and nearshore waters, this overlap exposes the giant manta ray in these locations to an increased likelihood of potential vessel strike. Yet, few instances of confirmed or suspected mortalities of giant manta ray attributed to vessel strike injury (e.g., via strandings) have been documented. This lack of documented mortalities could also be the result of other factors that influence carcass detection (i.e., wind, currents, scavenging, decomposition etc.). In addition, manta rays appear to be able to heal from wounds very quickly, while high wound healing capacity is likely to be beneficial for their long-term survival, the fitness cost of injuries and number vessel strikes occurring may be masked (McGregory et al., 2019).

Microplastics

Filter-feeding megafauna are particularly susceptible to high levels of microplastic ingestion and exposure to associated toxins due to their feeding strategies, target prey, and, for most, habitat overlap with microplastic pollution hotspots (Germanov et al. 2019). Giant manta rays are filter feeders, and, therefore can ingest microplastics directly from polluted water or indirectly through-contaminated planktonic prey (Miller and Klimovich 2017). The effects of ingesting indigestible particles include blocking adequate nutrient absorption and causing mechanical damage to the digestive tract. Microplastics can also harbor high levels of toxins and persistent organic pollutants, and introduce these toxins to organisms via ingestion. These toxins can bioaccumulate over decades in long-lived filter feeders, leading to a disruption of biological processes (e.g., endocrine disruption), and potentially altering reproductive fitness (Germanov et al. 2019). Jambeck et al. (2015) found that the Western and Indo- Pacific regions are responsible for the majority of plastic waste. These areas also happen to overlap with some of the largest known aggregations of giant manta rays. For example, in Thailand, where recent sightings data have identified over 288 giant manta rays (MantaMatcher 2016), mismanaged plastic waste is estimated to be on the order of 1.03 million tonnes annually, with up to 40% of this entering the marine environment (Jambeck et al. 2015). Approximately 1.6 million tonnes of mismanaged plastic waste is being disposed of in Sri Lanka, again with up to 40% entering the marine environment (Jambeck et al. 2015), potentially polluting the habitat used by the nearby Maldives aggregation of manta rays. While the ingestion of plastics is likely to negatively affect the health of the species, the levels of microplastics in manta ray feeding grounds and frequency of ingestion are presently being studied to evaluate the impact on these species (Germanov et al. 2019).

Mooring and Anchor Lines

Mooring and boat anchor line entanglement may also wound giant manta rays or cause them to drown (Deakos et al. 2011; Heinrichs et al. 2011). There are numerous anecdotal reports of giant manta rays becoming entangled in mooring and anchor lines (C. Horn, NMFS, unpublished data), as well as documented interactions encountered by other species of manta rays (C. Horn, NMFS, unpublished data). For example, although a rare occurrence, reef manta rays on occasion entangle themselves in anchor and mooring lines. Deakos (2010) suggested that manta rays become entangled when the line makes contact with the front of the head between the cephalic lobes, the animal's reflex response is to close the cephalic lobes, thereby trapping the rope between the cephalic lobes, entangling the manta ray as the animal begins to roll in an attempt to free itself. In Hawaii, on at least 2 occasions, a reef manta ray was reported to have died after entangling in a mooring line (A. Cummins, pers. comm. 2007, K. Osada, pers. comm. 2009; cited in Deakos (2011)). In Maui, Hawaii, Deakos et al. (2011) observed that 1 out of 10 reef manta rays had an amputated or disfigured non-functioning cephalic lobe, likely a result of line entanglement. Mobulid researchers indicate that entanglements may significantly affect the manta rays fitness (Braun et al. 2015; Convention on Migratory Species 2014; Couturier et al. 2012; Deakos et al. 2011; Germanov and Marshall 2014; Heinrichs et al. 2011). However, there is very little quantitative information on the frequency of these occurrences and no information on the impact of these injuries on the overall health of the species.

Climate Change Effects

Because giant manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the impacts of climate change compared to other sharks and rays (Chin et al. 2010). However, as giant manta rays frequently rely on coral reef habitat for important life history functions (e.g., feeding, cleaning) and depend on planktonic food resources for nourishment, both of which are highly sensitive to environmental changes (Brainard et al. 2011; Guinder and Molinero 2013), climate change is likely to have an impact on their distribution and behavior. Coral reef degradation from anthropogenic causes, particularly climate change, is projected to increase through the future. Specifically, annual, globally averaged surface ocean temperatures are projected to increase by approximately 0.7 °C by 2030 and 1.4 °C by 2060 compared to the 1986-2005 average (Intergovernmental Panel on Climate Change 2013), with the latest climate models predicting annual coral bleaching for almost all reefs by 2050 (Heron et al. 2016). Declines in coral cover have been shown to result in changes in coral reef fish communities (Jones et al. 2004) (Graham et al. 2008). Therefore, the projected increase in coral habitat degradation may potentially lead to a decrease in the abundance of fish that clean giant manta rays (e.g., *Labroides* spp., *Thalassoma* spp., and *Chaetodon* spp.) and an overall reduction in the number of cleaning stations available to manta rays within these habitats. Decreased access to cleaning stations may negatively affect the fitness of giant manta rays by hindering their ability to reduce parasitic loads and dead tissue, which could lead to increases in diseases and declines in reproductive fitness and survival rates.

Changes in climate and oceanographic conditions, such as acidification, are also known to affect zooplankton structure (size, composition, and diversity), phenology, and distribution (Guinder and Molinero 2013). As such, the migration paths and locations of both resident and seasonal aggregations of giant manta rays, which depend on these animals for food, may similarly be altered (Couturier et al. 2012). As research to understand the exact impacts of climate change on marine phytoplankton and zooplankton communities is still ongoing, the severity of this threat has yet to be fully determined (Miller and Klimovich 2017).

3.2.8 Oceanic Whitetip Shark

On January 30, 2018, NMFS published a final rule to list the oceanic whitetip shark (*Carcharhinus longimanus*) as a threatened species under the ESA, effective March 1, 2018 (83 FR 4153). The status review report of the oceanic whitetip shark (Young et al. 2016b) compiles the best available information on the status of the species as required by the ESA and assesses the current and future extinction risk for the species.

Species Description

The oceanic whitetip shark is a large open ocean apex predatory shark found in subtropical waters around the globe. This species belongs to the family Carcharhinidae and is classified as a requiem shark (containing migratory, live-bearing sharks of the warm seas) (Order Carcharhiniformes). The oceanic whitetip belongs to the genus Carcharhinus, which includes other pelagic species of sharks, such as the silky shark (*C. falciformis*) and dusky shark (*C. obscuras*), and is the only truly oceanic shark of its genus (Bonfil 2009).

The oceanic whitetip shark has a stocky build with a large rounded first dorsal fin and very long and wide paddle-like pectoral fins. The first dorsal fin is very wide with a rounded tip, originating just in front of the rear tips of the pectoral fins. The second dorsal fin originates over or slightly in front of the base of the anal fin. The species also exhibits a distinct color pattern of mottled white tips on its front dorsal, caudal, and pectoral fins with black tips on its anal fin and on the ventral surfaces of its pelvic fins. The head has a short and bluntly rounded nose and small circular eyes with nictitating membranes. The upper jaw contains broad, triangular serrated teeth, while the teeth in the lower jaw are more pointed and are only serrated near the tip. The body is grayish bronze to brown in color, but varies depending upon geographic location. The underside is whitish with a yellow tinge on some individuals. They usually cruise slowly at or near the surface with their huge pectoral fins conspicuously outspread, but can suddenly dash for a short distance when disturbed (Compagno 1984).

Distribution and Habitat Use

A geographical representation of the species range is provided by Last and Stevens (2009) (Figure 24). The oceanic whitetip shark is distributed worldwide in epipelagic tropical and subtropical waters between 30º North latitude and 35º South latitude (Baum et al. 2006). Although the oceanic whitetip can be found in decreasing numbers out to latitudes of 30° N and 35° S, with abundance decreasing with greater proximity to continental shelves, it has a clear preference for open ocean waters between 10° S and 10° N (Backus et al. 1956; Bonfil et al. 2008; Compagno 1984; Strasburg 1958). In the Western Atlantic, oceanic whitetips occur from Maine to Argentina, including the Caribbean and Gulf of Mexico. The oceanic whitetip shark is a highly migratory species of shark that is usually found offshore in the open ocean, on the outer continental shelf, or around oceanic islands in deep water, occurring from the surface to at least 152 meters (m) depth. Essential Fish Habitat (EFH) for the oceanic whitetip shark includes localized areas in the central Gulf of Mexico and Florida Keys, and depths greater than 200 m in the Atlantic (from southern New England to Florida, Puerto Rico and the U.S. Virgin Islands. The species can be found in waters between 15°C and 28°C, but it exhibits a strong preference for the surface mixed layer in water with temperatures above 20 °C, and is considered a surface-dwelling shark. It is however, capable of tolerating colder waters down to 7.75°C for short periods as exhibited by brief, deep dives into the mesopelagic zone below the thermocline (>200 m), presumably for foraging (Howey-Jordan et al. 2013; Howey et al. 2016). However, exposures to these cold temperatures are not sustained (Musyl et al. 2011; Tolotti et al. 2015) and there is some evidence to suggest the species tends to withdraw from waters below 15°C (e.g., the Gulf of Mexico in winter; Compagno 1984). The thermal preferences of oceanic whitetip sharks in conjunction with their reported range within 30° N and S suggest possible thermal barriers to inter-ocean basin movements around the southern tips of Africa and South America (Bonfil *et al.,* 2008; Musyl *et al.,* 2011; Howey-Jordan *et al.,* 2013; (Gaither et al. 2016)).

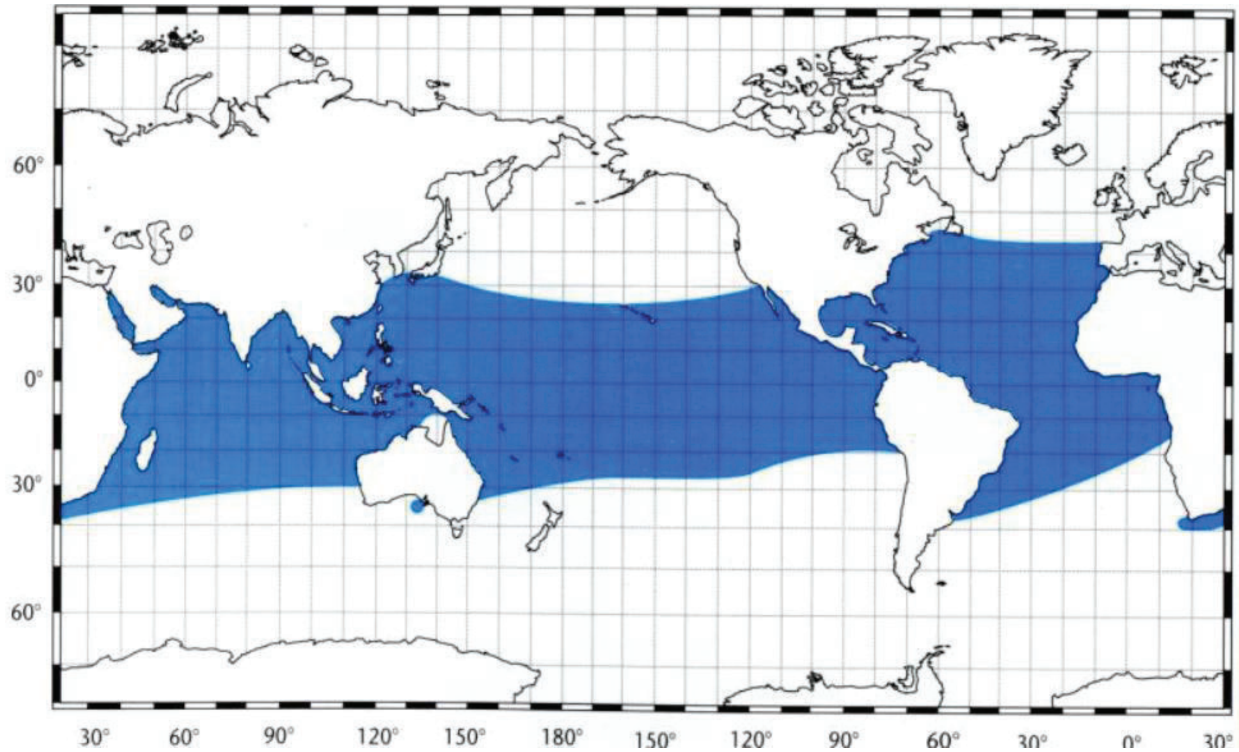

Figure 24. Geographic distribution of oceanic whitetip shark (Last and Stevens 2009)

Little is known about the movement or possible migration paths of the oceanic whitetip shark. Although the species is considered highly migratory and capable of making long distance movements, tagging data provides evidence that this species also exhibits a high degree of philopatry (i.e., site fidelity) in some locations. To date, there have been three tagging studies conducted on oceanic whitetip sharks in the Atlantic. In the Atlantic, young oceanic whitetip sharks have been found well offshore along the southeastern coast of the U.S., suggesting that there may be a nursery in oceanic waters over this continental shelf (Compagno 1984; Bonfil et al. 2008). In the southwestern Atlantic, the prevalence of immature sharks, both female and male, in fisheries catch data suggests that this area may serve as potential nursery habitat for the oceanic whitetip shark (Coelho et al. 2009; Frédou et al. 2015; Tambourgi et al. 2013; Tolotti et al. 2015). Juveniles seem to be concentrated in equatorial latitudes, while specimens in other maturational stages are more widespread (Tambourgi et al. 2013). Pregnant females are often found close to shore, particularly around the Caribbean Islands.

In the Atlantic Ocean, participants in the NMFS Cooperative Shark Tagging Program (CSTP) tagged 645 oceanic whitetips between 1962 and 2015, but only 8 were recaptured. Maximum time at liberty was 3.3 years, maximum distance traveled was 1,225 nmi (2,270 km), and maximum estimated speed was 17 nmi/day (32 km/day; Kohler *et al.,* (1998); NMFS unpublished data). These data show movements by juveniles from a variety of locations, including from the northeastern Gulf of Mexico to the East Coast of Florida, from the Mid-Atlantic Bight to southern Cuba, from the Lesser Antilles west into the central Caribbean Sea, from east to west along the equatorial Atlantic, and from off southern Brazil in a northeasterly direction (Kohler *et al.,* (1998); Bonfil *et al.,* (2008); see Figure 25). An immature female was also tagged in the waters between Cuba and Haiti and was recaptured the next day within 6 nmi (11 km) of the tagging location (NMFS unpublished data; see Figure 25). Additionally, an adult of unknown sex was tagged and recaptured three years apart in the vicinity of Cat Island, Bahamas (NMFS unpublished data; see Figure 25 below).

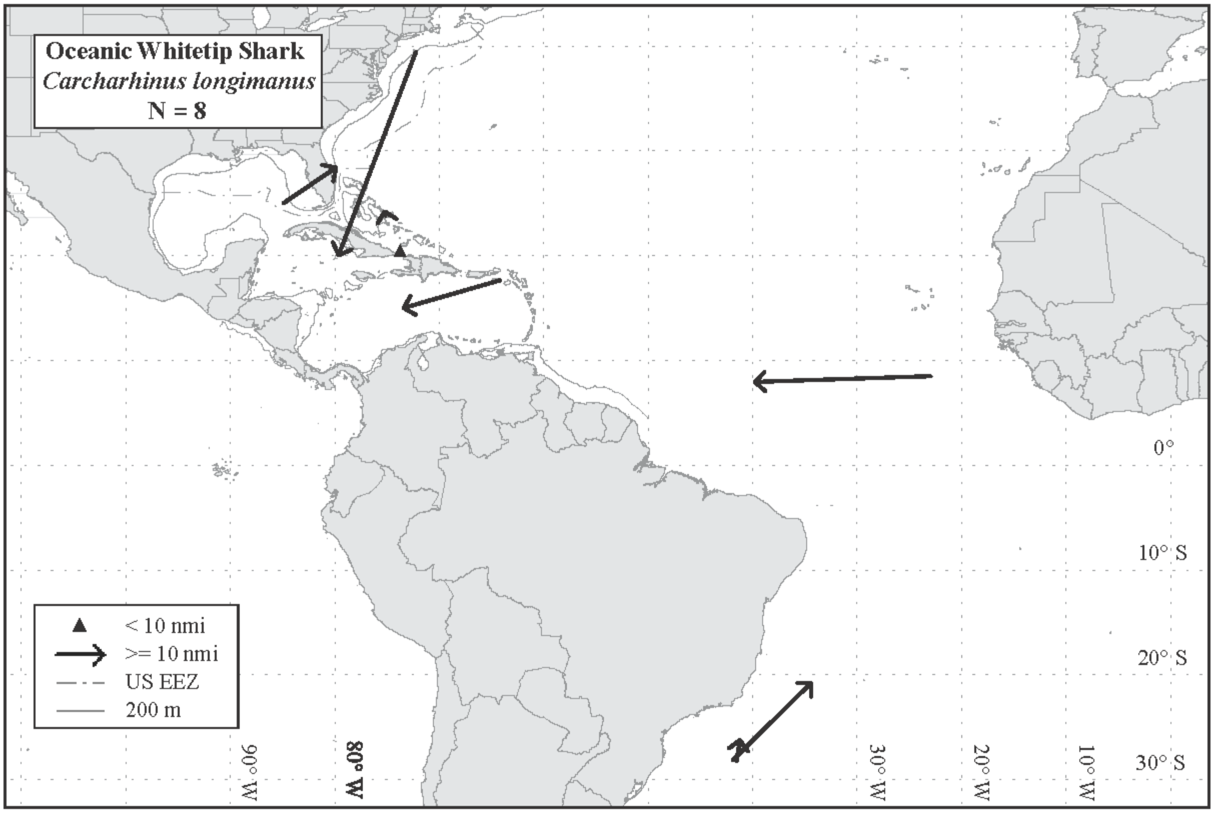

Figure 25. Recapture distribution for the oceanic whitetip shark from the NMFS Co-Operative Shark Tagging Program during 1962-1993 and NMFS Unpublished Data.

In the Gulf of Mexico, a satellite tagged oceanic whitetip shark moved a straight-line distance of 238 km from southeast Louisiana to the edge of the continental shelf about 300 km north of the Yucatan Peninsula. During the track, the shark rarely dove below 150 m staying above the thermocline, and only one dive to 256 m was recorded. The most frequently occupied depth during the entire track was 25.5-50 m (49.8% total time) and temperature was 24.05-26 °C (44.7% total time) (Carlson and Gulak 2012). More recently, a study from Cat Island, Bahamas tagged and tracked 11 mature oceanic whitetip sharks (10 females, 1 male). Individuals tagged at Cat Island stayed within 500 km of the tagging site for ~30 days before scattering across 16,422 km2 of the western North Atlantic (Howey-Jordan *et al*. 2013). Times at liberty ranged from 30-245 days, after which the largest movement by an individual from the tagging site ranged from 290–1,940 km. Individuals moved to several different destinations thereafter (e.g., the northern Lesser Antilles, the northern Bahamas, and north of the Windward Passage (the strait between Cuba and Haiti)), with many returning to the Bahamas after ~150 days. Howey-Jordan *et al.* (2013) found generally high residency times of oceanic whitetips in the Bahamas Exclusive Economic Zone (mean = 68.2% of time). Similar to the tagging study in the Pacific by Musyl *et* *al.,* (2011), oceanic whitetip sharks in the Bahamas spent 99.7% of their time in waters shallower than 200 m and did not show differences mean depths between day and night, with average day and night temperatures of 26.26±0.003oC and 26.23±0.003oC, respectively. According to Howey-Jordan et al. (2013):

> There was a positive correlation between daily sea surface temperature (SST) and mean depth occupied (i.e., as individuals experienced warmer SST, likely resulting from seasonal sea surface warming or migration to areas with warmer SST, mean daily depth increased, suggesting possible behavioral thermoregulation. All individuals made short duration (mean=13.06 minutes) dives into the mesopelagic zone (down to 1,082 m and 7.75◦C), which occurred significantly more often at night.

These tracking data also suggest that oceanic whitetip sharks exhibit site fidelity to Cat Island, Bahamas, although the reasons for this are still unclear. NMFS CSTP data (discussed earlier) from an adult oceanic whitetip, tagged and recaptured three years later in this area, provides supporting evidence of site fidelity to the waters around Cat Island. This information is important given the characterization of this species as highly migratory (Howey-Jordan *et al.,* 2013) (Figure 26).

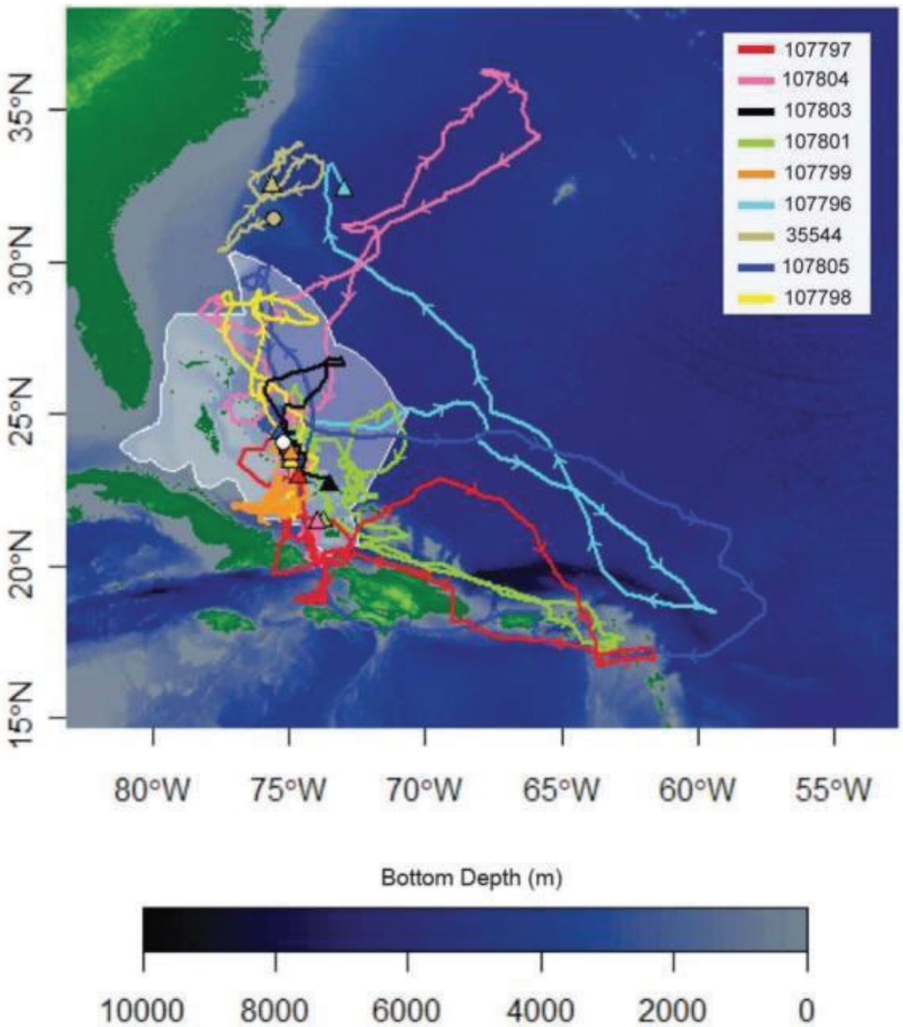

Figure 26. Map with bottom depth showing filtered tracks for nine oceanic whitetip sharks equipped with Standard Rate tags. Colored lines represent tracks from individuals (listed by tag ID) (Howey-Jorden et al. 2013)

For more information on oceanic whitetip distribution, see Young et al. (2016).

Life History Information

The oceanic whitetip shark gives birth to live young (i.e., "viviparous"). Their reproductive cycle is thought to be biennial, giving birth on alternate years, after a lengthy 10–12 month gestation period. The number of pups in a litter ranges from 1 to 14 (mean = 6), and a positive correlation between female size and number of pups per litter has been observed, with larger sharks producing more offspring (Bonfil et al. 2008; Compagno 1984; IOTC 2014; Seki et al. 1998). Age and length of maturity estimates are slightly different depending on geographic location. In the Southwest Atlantic, age and length of maturity in oceanic whitetips was estimated to be 6–7 years and 180–190 cm total length (TL), respectively, for both sexes (Lessa et al. 1999).

Historically, the maximum length effectively measured for the oceanic whitetip was 350 cm TL (Bigelow and Schroder 1948 cited in Lessa et al. 1999), with ‘‘gigantic individuals’’ perhaps reaching 395 cm TL (Compagno 1984), though Compagno’s length seems to have never been measured (Lessa et al. 1999). In contemporary times, Lessa et al. (1999) recorded a maximum size of 250 cm TL in the Southwest Atlantic, and estimated a theoretical maximum size of 325 cm TL (Lessa et al. 1999), but the most common sizes are below 300 cm TL (Compagno 1984). The oceanic whitetip has an estimated maximum age of 17 years, with confirmed maximum ages of 12 and 13 years in the North Pacific and South Atlantic, respectively (Seki et al. 1998; Lessa et al. 1999). However, other information from the South Atlantic suggests the species likely lives up to ~20 years old based on observed vertebral ring counts (Rodrigues et al. 2015). Growth rates (growth coefficient, K) have been estimated similarly for both sexes and range from 0.075—0.099 in the Southwest Atlantic to 0.0852–0.103 in the North Pacific (Joung et al. 2016; Lessa et al. 1999; Seki et al. 1998). Using life history parameters from the Southwest Atlantic, (Cortés et al. 2010; Cortés et al. 2012) estimated productivity of the oceanic whitetip shark, determined as intrinsic rate of population increase (r), to be 0.094–0.121 per year (median). Overall, the best available data indicate that the oceanic whitetip shark is a longlived species (at least 20 years) and can be characterized as having relatively low productivity.

To date, only two studies have been conducted on the genetics and population structure of the oceanic whitetip shark, which suggest there may be some genetic differentiation between various populations of the species. Overall, the data showing population structure within the Atlantic relies solely on mitochondrial DNA and does not reflect male mediated gene flow. Thus, while the current data supports three maternal populations within the Atlantic, information regarding male mediated gene flow would provide an improved understanding of the fine-scale genetic structuring of oceanic whitetip in the Atlantic. On the other hand, both mitochondrial DNA and nuclear microsatellite data analyses support at least two global genetic stocks. However, the data from these studies are preliminary, and it is likely that additional population structure within and between oceans will be discovered with additional samples and analyses.

Oceanic whitetip sharks are high trophic-level predators in open ocean ecosystems feeding mainly on teleosts and cephalopods (Backus et al. 1956; Bonfil et al. 2008), but studies have also reported that they consume sea birds, marine mammals, other sharks and rays, molluscs, crustaceans, and even garbage (Compagno 1984; Cortés 1999). Backus et al. (1956) recorded various fish species in the stomachs of oceanic whitetip sharks, including blackfin tuna, barracuda, and white marlin. Based on the species’ diet, the oceanic whitetip has a high trophic level, with a score of 4.2 out of a maximum 5.0 (Cortés 1999). The available evidence also suggests that oceanic whitetip sharks are opportunistic feeders.

Status and Population Dynamics

Oceanic whitetip sharks can be found worldwide, with no present indication of a range contraction. While a global population size estimate or trend for the oceanic whitetip shark is currently unavailable, numerous sources of information, including the results of a recent stock assessment and several other abundance indices (e.g., trends in occurrence and composition in fisheries catch data, CPUE, and biological indicators) were available to infer and assess current regional abundance trends of the species. Given the available data, and the fact that the available assessments were not conducted prior to the advent of industrial fishing (and thus not from virgin biomass), the exact magnitude of the declines and current abundance of the global population are unknown. The oceanic whitetip shark was historically one of the most abundant and ubiquitous shark species in tropical seas around the world; however, numerous lines of evidence suggest declines greater than 70-80% in most areas throughout its range, and this species likely continues to experience abundance declines of varying magnitude globally.

In the Northwest Atlantic, the oceanic whitetip shark was described historically as widespread, abundant, and the most common pelagic shark in the warm parts of the North Atlantic (Backus et al. 1956). Recent information, however, suggests the species is now relatively rare in this region.

Several studies have been conducted in this region to determine trends in abundance of various shark species, including the oceanic whitetip shark, and these studies have shown significant declines in abundance. The proposed listing rule provides more detail on the varying estimates on the severity of the declines (81 FR 96304; December 29, 2016). Relative abundance of oceanic whitetip shark may have stabilized in the Northwest Atlantic since 2000 and in the Gulf of Mexico/Caribbean since the late 1990s at a significantly diminished abundance (Young et al. 2016b).

Threats

Currently, the most significant threat to oceanic whitetip sharks is mortality in commercial fisheries, largely driven by demand of the international shark fin trade, bycatch related mortality, as well as illegal, unreported, and unregulated (IUU) fishing. Although generally not targeted, oceanic whitetip sharks are frequently caught as bycatch in many fisheries, including pelagic longline fisheries targeting tuna and swordfish, purse seine, gillnet, and artisanal fisheries. Oceanic whitetip sharks are also a preferred species for their large, morphologically distinct fins, as they obtain a high price in the Asian fin market. The oceanic whitetip shark's vertical and horizontal distribution significantly increases its exposure to industrial fisheries, including pelagic longline and purse seine fisheries operating within the species' core tropical habitat throughout its global range.

In addition to declines in oceanic whitetip catches throughout its range, there is also evidence of declining average size over time in some areas, and is a concern for the species' status given evidence that litter size is positively correlated with maternal length. Such extensive declines in the species' global abundance and the ongoing threat of overutilization, the species' slow growth and relatively low productivity, makes them generally vulnerable to depletion and potentially slow to recover from overexploitation. Related to this, the low genetic diversity of oceanic whitetip is also cause for concern and a viable risk over the foreseeable future for this species. Loss of genetic diversity can lead to reduced fitness and a limited ability to adapt to a rapidly changing environment. The biology of the oceanic whitetip shark indicates that it is likely to be a species with low resilience to fishing and minimal capacity for compensation (Rice and Harley 2012).

Available information does not indicate that destruction, modification or curtailment of the species' habitat or range, disease or predation, or other natural or manmade factors are operative threats on this species (81 FR 96304; December 29, 2016).

3.2.9 Sperm Whale

Sperm whales were first listed under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973 [(35 FR 18319 1970), December 2, 1970]. The primary cause of the population decline that precipitated ESA listing was commercial whaling for ambergris and spermaceti in the eighteenth, nineteenth, and twentieth centuries. Hunting of sperm whales by commercial whalers declined in the 1970s and 1980s, and virtually ceased with the implementation of a moratorium against whaling by the International Whaling Commission (IWC) in 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). The IWC estimates that nearly 250,000 sperm whales were killed worldwide in whaling activities between 1800 and 1900. From 1910 to 1982, nearly 700,000 sperm whales were killed worldwide by whaling activities (IWC Statistics 1959-1983). A compilation of all whaling catches in the North Atlantic north of 20ºN from 1905 onward gave totals of 28,728 males and 9,507 females (NMFS 2010). Sperm whales are also protected under the Marine Mammal Protection Act (MMPA) and also listed in Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), meaning that commercial trade in products of sperm whales is prohibited.

Species Description

The sperm whale (*Physeter microcephalus*, Linnaeus 1758), occurs in all oceans of the world. Sperm whales are perhaps the most widely distributed mammal on earth. It is the largest of the toothed whales, reaching a length of 60 ft (18.3 m) in males and 40 ft (12.2 m) in females (Odell 1992). Sperm whales are distributed throughout most oceanic areas, but are found in deeper waters seaward of the continental shelf. Deep water is required so they can make prolonged, deep dives to locate prey, breed, and nurse their young. In general, females and immature sperm whales appear to be restricted in range, whereas males are found over a wider range and do make occasional movements across and between ocean basins (Dufault et al. 1999). Stable, long-term associations among related and unrelated females form the core units of sperm whale societies (Christal et al. 1998). Females and juveniles form groups that are generally within tropical and temperate latitudes between 50°N and 50°S, while the solitary adult males can be found at higher latitudes between 75ºN and 75ºS (Reeves and Whitehead 1997). The home ranges of individual females seem to span distances of approximately 1,000 km (Best 1979; Dufault and Whitehead 1995). Although there is strong evidence for geographic, matrilineal structuring in sperm whales, there is no evidence these management stocks represent distinct populations of whales.

The Recovery Plan (NMFS 2010) identifies recovery criteria geographically across three ocean basins: the Atlantic Ocean/Mediterranean Sea, the Pacific Ocean, and the Indian Ocean. This geographic division by basin is due to the wide distribution of sperm whales and presumably little movement of whales between ocean basins. For management purposes under the MMPA, sperm whales inhabiting U.S. waters have been divided into 5 stocks: (1) the California-Oregon-Washington Stock, (2) the North Pacific (Alaska) Stock, (3) the Hawaii Stock, 4) the Northern Gulf of Mexico Stock, (5) and the North Atlantic Stock. In the Gulf of Mexico, sperm whales are the most common large cetacean seaward of the continental shelf (Davis et al. 1998; Jefferson and Schiro 1997; Mullin et al. 1991; Mullin and Fulling 2004; Mullin et al. 1994; Weller et al. 2000; Wursig et al. 2000). Sperm whales in the Gulf of Mexico are not evenly distributed, showing greater densities in areas associated with oceanic features that provide the best foraging opportunities (Figure 27).

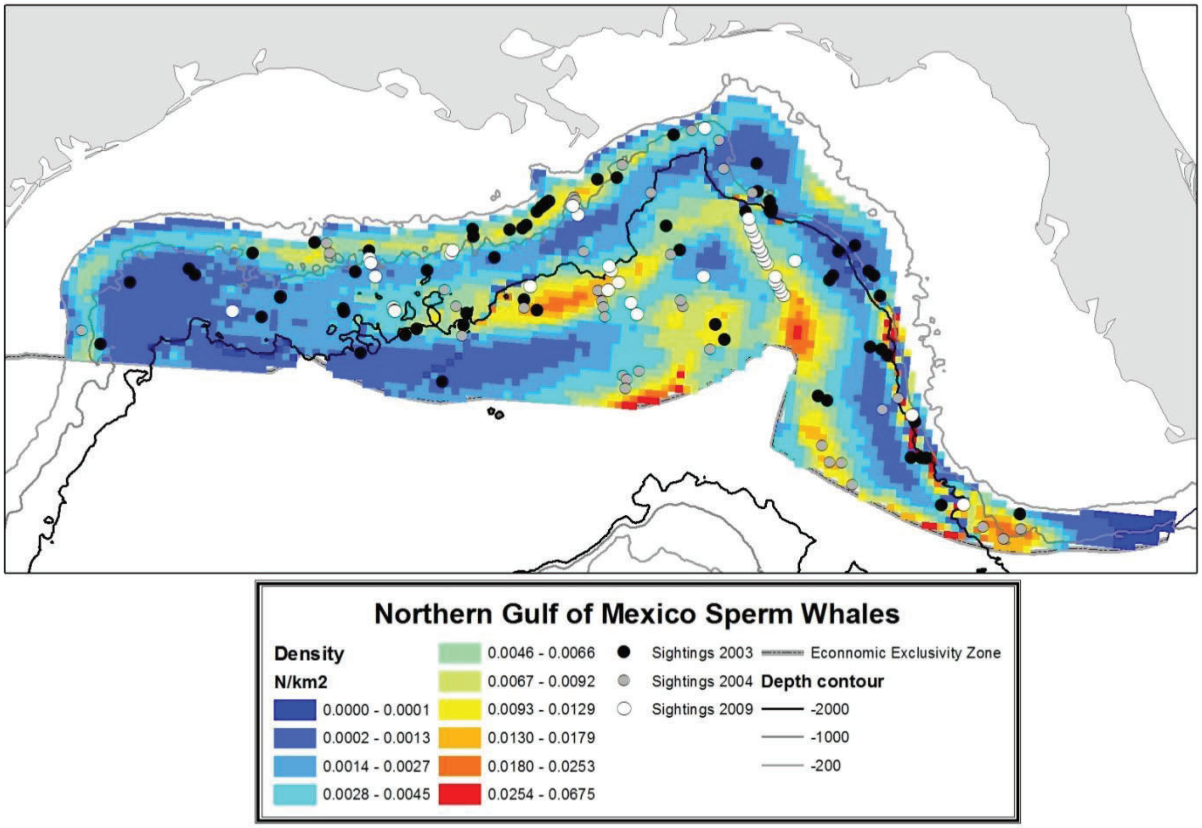

Figure 27. Sperm whale sightings and predicted densities in the Gulf of Mexico

Life History Information

The social organization of sperm whales, and with most other mammals, is characterized by females remaining in the geographic area in which they were born and males dispersing more broadly. Females group together and raise young. For female sperm whales, remaining in the region of birth can include very large oceanic ranges over which the whales need to successfully forage and nurse young whales. Male sperm whales are mostly solitary and disperse more widely and can mate with multiple female populations throughout a lifetime.

Female and immature sperm whales of both sexes are found in more temperate and tropical waters throughout the year. Maturing males will leave the female groups and form loose aggregations of bachelor schools. As the males grow older, they separate from the bachelor schools and remain solitary most of the year (Best 1979). Adult males visit female groups of whales only to breed. Large males have been sighted on occasion and are believed to enter the Gulf of Mexico for short periods to breed. Therefore, the Gulf of Mexico population is comprised of the year-round presence of females, calves, and juvenile whales. Female sperm whales attain sexual maturity at the mean age of 8 or 9 years. The mature females ovulate April through August in the Northern Hemisphere. Maturation in males usually begins in this same age interval as females, but males have a prolonged puberty and attain sexual maturity at between age 12 and 20. Males may require another 10 years to become large enough to successfully compete for breeding rights (Kasuya 1991). During this season of ovulating females, 1 or more large mature bulls temporarily join each breeding school. In the North Atlantic Ocean, the peak breeding season for sperm whales occurs during the spring (March/April to June), although some mating activity continues throughout the summer. In the South Atlantic Ocean, the peak breeding season is presumed to occur in the austral spring. During mating seasons, mature males in their late twenties and older rove among groups of females. Because females within a group often become reproductively active at the same time, the male need not remain with them for an entire season to achieve maximal breeding success (Best and Butterworth 1980) and their association with a female group can be as brief as several hours. Gestation lasts well over a year, with credible estimates of the normal duration ranging from 15 months to more than a year and a half. A single calf is born at a length of about 13 ft (4 m), after a 15-16 month gestation period. Female sperm whales rarely become pregnant after the age of 40 (Whitehead 2003). Females assist each other in the care of offspring, guarding of young at the surface while mothers dive (Whitehead 1996). Females even have been observed nursing valves other than their own (Reeves and Whitehead 1997). Calves are nursed for 2-3 years (in some cases, up to 13 years), and the calving interval is estimated to be about 4-7 years (Kasuya 1991).

The age distribution of the sperm whale population is unknown, but they are believed to live at least 60 years (Rice 1989). Potential sources of natural mortality in sperm whales include killer whale predation and the papilloma virus (Lambertsen et al. 1987). Sperm whales may also be "harassed" by pilot whales (*Globicephala spp*.) and false killer whales (*Pseudorca crassidens*), but most "attacks" by these species are probably unsuccessful (Palacios and Mate 1996; Weller et al. 1996). Very little is known about the role of disease in the natural mortality of sperm whales (Lambertsen et al. 1987). Only 2 naturally occurring diseases that are likely to be lethal have been identified in sperm whales: myocardial infarction associated with coronary atherosclerosis, and gastric ulceration associated with nematode infection (Lambertsen et al. 1987). There have been 14 individual sperm whale strandings reported in the Gulf of Mexico from 2002-2012. The rate of stranding in the Gulf of Mexico is not unusual and does not suggest mortality levels that are not sustainable to the population.

Cephalopods (i.e., squid, octopi, cuttlefishes, and nautili) are the main component of sperm whale diets. The ommastrephids, onychoteuthids, cranchids, and enoploteuthids are the cephalopod families that are numerically important in the diet of sperm whales in the Gulf of Mexico (Davis et al. 2002). Other populations, especially mature males in higher latitudes, are known to feed on significant quantities of large demersal and mesopelagic sharks, skates, and bony fishes (Clarke 1962; Clarke 1979). Sperm whales consume about 3.0-3.5% of their body weight per day (Lockyer 1981). Sperm whales undergo deep foraging dives to find prey. Descent rates are approximately 1.7 m/s and nearly vertical (Goold and Jones 1995). Dive depth may be dependent upon temporal variations in prey location in the water column.

Typical foraging dives last 40 minutes to depths of about 1,300 ft (400 m), followed by approximately 8 minutes of resting at the surface (Gordon 1987; Papastavrou et al. 1989). Nonetheless, dives of over 2 hours and deeper than 2 miles (3.3 km) have been recorded (Clarke 1976); individuals may spend extended periods of time at the surface to recover.

The disproportionately large head of the sperm whale is an adaptation to produce acoustic signals (Cranford 1992; Norris and Harvey 1972). Sperm whales locate prey by echolocation clicks while in a deep dive pattern, and also produce vocalizations while resting at the surface. The function of vocalizations is relatively well-studied (Goold and Jones 1995; Weilgart and Whitehead 1997). Long series of monotonous, regularly spaced clicks and closely spaced clicks are produced for echolocation and are associated with feeding and prey capture. Clicks produced by sperm whales (and presumably heard by them) are in the range of about 0.1-20 kHz (Goold and Jones 1995; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997), up to 30 kHz, often with most of the energy in the 2-4 kHz range (Watkins 1980). Clicks have source levels estimated at 171 dB re: 1 μPa (Levenson 1974).

Sperm whales also utilize unique stereotyped click sequences called "codas" (Adler-Fenchel 1980; Mullins et al. 1988; Watkins et al. 1985a; Watkins and Schevill 1977). Codas may convey information about the age, sex, and reproductive status of the sender (Weilgart and Whitehead 1988), and may maintain social cohesion with the group (Weilgart and Whitehead 1993). Sperm whales show regional differences in coda patterns (Weilgart and Whitehead 1997). Sperm whales have been categorized as a cetacean in the mid-frequency functional hearing group in the range of 150-160 kHz and can hear wide variety of sounds in the ocean environment.

Population Size and Genetic Variability

The best estimate for the current worldwide abundance of sperm whale is estimated between 300,000-450,000 individuals (Whitehead 2002). The abundance of sperm whales in the Atlantic Ocean is estimated at 90,000-134,000 individuals and 763 whales in the northern Gulf of Mexico. On a global scale, no genetic differences have been found in the nuclear DNA (nDNA) (bi-parentally inherited) between individuals sampled in different ocean basins with some differences found in mitochondrial DNA (mtDNA) (maternally-inherited) sequences (Lyrholm et al. 1999). In general, results tend to find low genetic differentiation of nDNA among sperm whales in different ocean basins and little differentiation of mtDNA within ocean basin stocks, with the exception of some semi-enclosed basins such as the Mediterranean Sea and Gulf of Mexico (Dillon et al. 1997; Lyrholm and Gyllensten 1998; Mesnick et al. 1999) Bond (1999); (Engelhaupt 2004; Lyrholm et al. 1999). Based on over 2,473 tissue samples and 1,038 mtDNA sequences from a global consortium of investigators, 28 haplotypes have been identified worldwide, defined by 24 variable sites (Mesnick et al. 2005). Three common haplotypes dominated the sequencing and made up 82% of the total. This dominance by a few haplotypes indicates broad reproductive mixing of genetic material. Mitochondrial DNA evidence in the Gulf of Mexico suggests population structuring based on genetic material inherited from mothers. Regional structuring is also supported by satellite tracking data suggesting that most females establish home ranges within the Gulf of Mexico basin, and their site fidelity has resulted in maternally related groups of females and young whales in this region.

Current Threats

Since the ban on nearly all hunting of sperm whales, levels of anthropogenic mortality and injury have been comparatively low (Perry et al. 1999; Waring et al. 2002). Sperm whale numbers were drastically reduced due to whaling. Although the worldwide population has not recovered, sperm whales numbers no longer appear to be in decline. Two particular aspects of the sperm whale's reproductive biology are relevant to management and recovery of the species. First, the maximal rate of increase from reproduction is very low, perhaps no more than 1% or 2% of the population per year. Second, selective killing of large males by historical whaling could have a residual effect of reducing reproductive rates (Whitehead et al. 1997).

Current threats to sperm whales include ship strikes and entanglements in fishing gear. Other threats to sperm whales include disturbance by man-made noise, for example from seismic surveys, and this threat is heightened in areas of oil and gas activities or where shipping activity is high. NMFS' Recovery Plan for Sperm Whales (NMFS 2010) identified 4 main categories of threats to the recovery of sperm whales in the Atlantic Ocean: (1) vessel interactions, (2) incidental capture in fishing gear, (3) habitat degradation, and (4) military operations.

Vessels affect sperm whales via collisions and vessel noise. Sperm whales spend periods of up to 10 minutes "rafting" at the surface between deep dives (Jaquet et al. 1998). This could make them exceptionally vulnerable to ship strikes. Studies on the behavioral of sperm whales around whale watching boats suggest sperm whales change their diving and acoustic behavior in response to boats, but following frequent exposure, they become increasingly tolerant or habituated to the presence of vessels (Gordon et al. 1992; Markowitz 2011).

Incidental entrapment and entanglement in fishing gear, especially gillnets set in deep water for pelagic fish (e.g., sharks, billfish, tuna), is of potential concern. In U.S. east-coast waters, two incidents were reported between 1990 and 1995, both on Georges Bank. In 1990, one whale was found entangled and was released in "injured" condition. In 1995, another was found, also injured, and released while still carrying gear (Waring et al. 1997). Based on observer data from the drift-net fishery, mortality of sperm whales between 1989 and 1995 ranged from 0 to 4.4 (CV 1.77) per year (Waring et al. 1997). A single nonlethal interaction between sperm whales and the longline fishery have been recorded in the U.S. Atlantic Gulf of Mexico. Only 1 stranded sperm whale has shown signs of human interaction.

The accumulation of stable pollutants (e.g. heavy metals, polycholorobiphenyls [PCBs], chlorinated pesticides [DDT, DDE, etc.], and polycyclic aromatic hydrocarbons [PAHs]) is of concern. The potential impact of coastal pollution may be an issue for this species in portions of its habitat, though little is known regarding the effect pollutants may have on individuals. Because sperm whales feed at high trophic levels and store the chemicals in their blubber, they are susceptible to chemical pollution. Sperm whales could potentially pass these chemicals to their offspring in their milk (Whitehead 2003). A population sensitivity analysis for the Gulf of Mexico sperm whales showed that if toxins, such as those found in oil spills, reduce the survivorship rate of the mature female sperm whales by as little as 2.2% or the survivorship rate of mothers by 4.8%, the growth rate of the population would drop to a level that would result in a decline in the size of that population (Chiquet et al. 2013).

Marine debris may be ingested by sperm whales as is the case with many marine animals. Debris entrained in the deep scattering layer where sperm whales feed could be mistaken for prey and incidentally ingested. Man-made noise and offshore energy development may also be adversely affecting habitat quality. Because of their apparent role as important predators of mesopelagic squid and fish, changing the abundance of sperm whales should affect the distribution and abundance of other marine species. Conversely, changes in the abundance of mesopelagic squid and fish from recently developed targeted fisheries could affect the distribution of sperm whales.

Sperm whales are potentially affected by military operations in a number of ways. Whales can be struck by vessels and disturbed by sonar and other artificial sounds. Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins et al. 1985b; Watkins and Schevill 1975). They also stop vocalizing for brief periods when codas are being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

3.3 Potential Routes of Effect Not Likely to Adversely Affect Critical Habitat

Construction activities as well as vessel traffic related to construction, operation and eventual decommissioning of the proposed DWP projects will occur in loggerhead sea turtle designated critical habitat (LOGG-S-2, Gulf of Mexico *Sargassum*). The physical and biological feature of *Sargassum* habitat is described as developmental and foraging habitat for young loggerheads where surface waters form accumulations of floating material, especially *Sargassum*. 50 CFR 226.223(b)(5). The following primary constituent elements (PCEs) are present in LOGG-S-2:

(i) Convergence zones, surface-water downwelling areas, the margins of major boundary currents (Gulf Stream), and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for the optimal growth of *Sargassum* and inhabitance of loggerheads;

(ii) *Sargassum* in concentrations that support adequate prey abundance and cover;

(iii) Available prey and other material associated with *Sargassum* habitat including, but not limited to, plants and cyanobacteria and animals native to the *Sargassum* community such as hydroids and copepods; and

(iv) Sufficient water depth and proximity to available currents to ensure offshore transport (out of the surf zone), and foraging and cover requirements by *Sargassum* for post-hatchling loggerheads, i.e., >10 m depth.

50 CFR 226.223(b)(5)(i)-(iv).

The PCEs that may be affected by the proposed actions include: (ii) *Sargassum* in concentrations that support adequate prey abundance and cover, and (iii) Available prey and other material associated with *Sargassum* habitat including, but not limited to, plants and cyanobacteria and animals native to the *Sargassum* community such as hydroids and copepods. Construction barges, support vessels, and oil transport vessels associated with the proposed actions are not anticipated to scatter *Sargassum* mats or harm organisms in the *Sargassum* to the point of affecting the functionality of the loggerhead critical habitat PCEs. The wakes and surface water disruptions associated with these vessels may temporarily disturb a *Sargassum* mat (i.e., for a few minutes up to a few hours). However, any potential disturbance would be insignificant, as it would not be expected to result in measurable effects to the distribution, size, or composition of mats or their ability to support loggerheads or their prey resources.

3.4 Status of Critical Habitat Likely to be Adversely Affected

On September 22, 2011, NMFS and USFWS jointly published a final rule revising the listing of loggerhead sea turtles from a single worldwide threatened species to nine DPSs. One of those nine DPSs, the NWA DPS, is located in NMFS SERO jurisdiction. At the time the final listing rule was developed, we lacked comprehensive data and information necessary to identify and describe physical or biological features (PBFs) of the terrestrial and marine habitats. As a result, we found designation of critical habitat to be "not determinable" (see 16 U.S.C. 1533(b)(6)(C)(ii)) at the time of listing. In the final rule we stated that we would consider designating critical habitat in future rulemakings after a critical habitat review team was convened to assess and evaluate potential critical habitat areas for the DPSs in U.S. waters. The Services published a proposed rule (78 FR 43005) to designate critical habitat for the threatened Northwest Atlantic Ocean DPS on July 18, 2013, followed by the final rule on July 10, 2014 (79 FR 39855).

The final rule designates 38 marine areas within portions of the northwestern Atlantic Ocean and the Gulf of Mexico as critical habitat (Figure 28). Each of these areas consists of one or a combination of the following habitat types: (1) nearshore reproductive habitat (directly off high density nesting beaches out to 1 mile (1.6 km)), (2) wintering habitat, (3) breeding habitat, (4) constricted migratory corridors, and (5) *Sargassum* habitat. We determined that these habitat types support key life history phases of the loggerhead sea turtle and are thus essential to the conservation of the species. To further define critical habitat, we identified the physical and biological features (PBFs) and the primary constituent elements (also referred to as "essential features") of the habitat that are essential for the conservation of the NWA DPS (Table 9).

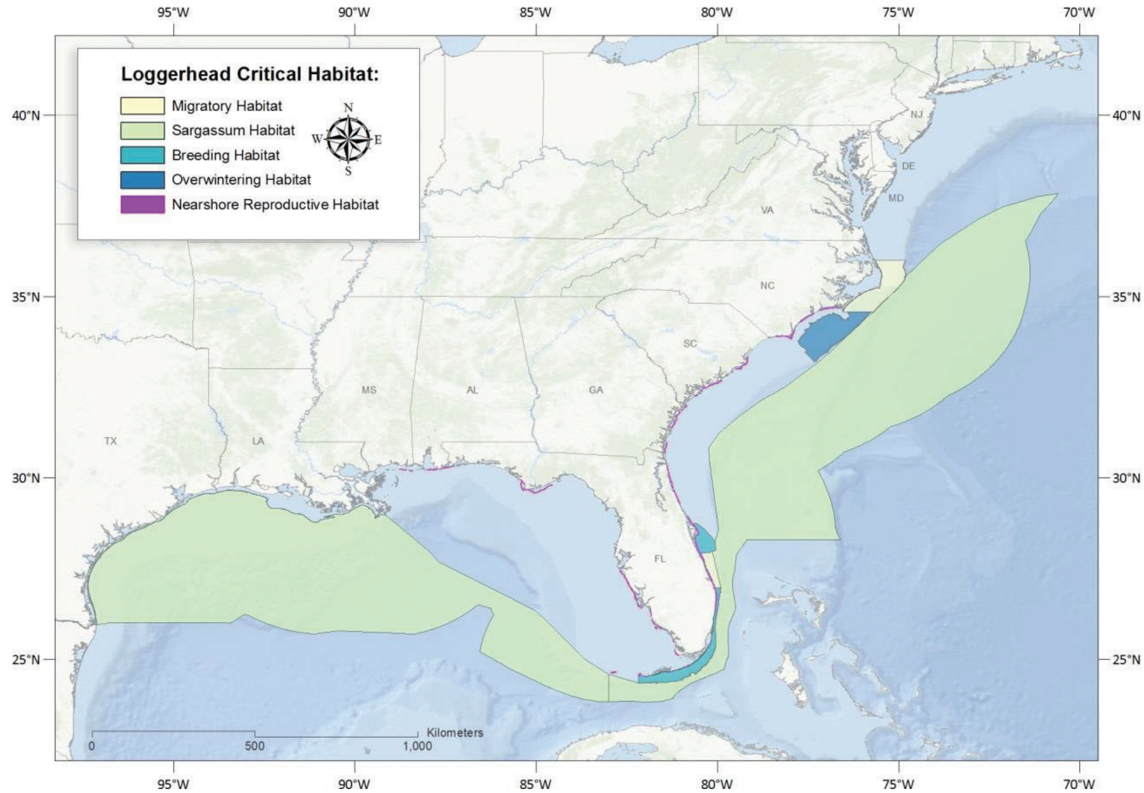

Figure 28. Distribution of critical habitat for the NWA DPS of loggerhead sea turtles

Table 9. Description of critical habitat for the NWA DPS of loggerhead sea turtles

Habitat Type	Physical & Biological Feature	Primary Constituent Elements	Unit Numbers
Nearshore Reproductive	Portion of nearshore waters adjacent to nesting beaches that are used by hatchlings to egress to the open-water environment as well as by nesting females to transit between beach and open water during the nesting season	• Waters directly off the highest density nesting beaches to 1.6 km (1 mile) offshore • Waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water • Waters with minimal manmade structures that could promote predators (e.g., submerged offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents	LOGG-N-1 through LOGG-N-36
Winter	Warm water habitat south of Cape Hatteras near the western edge of the Gulf Stream used by concentration of	• Water temperatures above 10° C during colder months of November through April	LOGG-N-1 LOGG-N-2

Habitat Type	Physical & Biological Feature	Primary Constituent Elements	Unit Numbers
	juveniles and adults during the winter months	• Continental shelf waters in proximity to the western boundary of the Gulf Stream • Water depths between 20 and 100 meters	
Breeding	Areas with high concentrations of both male and female adult individuals during the breeding season	• Concentrations of reproductive males and females • Proximity to primary Florida migratory corridor • Proximity to Florida nesting grounds	LOGG-N-17 LOGG-N-19
Constricted Migratory	High use migratory corridors that are constricted (limited in width) by land on one side and the edge of the continental shelf and Gulf Stream on the other side	• Constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways • Passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas	LOGG-N-1, LOGG-N-17, LOGG-N-18, LOGG-N-19
Sargassum	Developmental and foraging habitat for young loggerheads where surface waters form accumulations of floating material, especially *Sargassum*.	• Convergence zones, surface-water downwelling areas, and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for the optimal growth of *Sargassum* and inhabitance of loggerheads • *Sargassum* in concentrations that support adequate prey abundance and cover • Available prey and other material associated with *Sargassum* habitat such as, but not limited to, plants and cyanobacteria and animals endemic to the *Sargassum* community such as hydroids and copepods	LOGG-S-1 LOGG-S-2

Critical Habitat Unit(s) Impacted by the Proposed Actions
The proposed actions will occur within the Gulf of Mexico and encompass loggerhead critical habitat Unit LOGG-S-02. Unit LOGG-S-02 only contains *Sargassum* habitat. The location of LOGG-S-02 is described below and the PCEs of this *Sargassum* habitat type are detailed in Table 9 above.

- LOGG-S-2—Gulf of Mexico *Sargassum* (Figure 29). The northern and western boundaries of the unit follow the 10 m depth contour starting at the mouth of South Pass of the Mississippi River proceeding west and south to the outer boundary of the U.S. [Exclusive Economic Zone] EEZ. The southern boundary of the unit is the U.S. EEZ from the 10 m depth contour off of Texas to the Gulf of Mexico-Atlantic border (83° W. long.). The eastern boundary follows the 10 m depth contour from the mouth of South Pass of the Mississippi River at 28.97° N. lat., 89.15° W. long., in a straight line to the northernmost boundary of the Loop Current (28° N. lat., 89° W. long.) and along the eastern edge of the Loop Current roughly following the velocity of 0.101-0.20 m/second as depicted by Love (2013) using the Gulf of Mexico summer mean sea surface currents from 1993-2011, to the Gulf of Mexico-Atlantic border (24.58° N. lat., 83° W. long.).

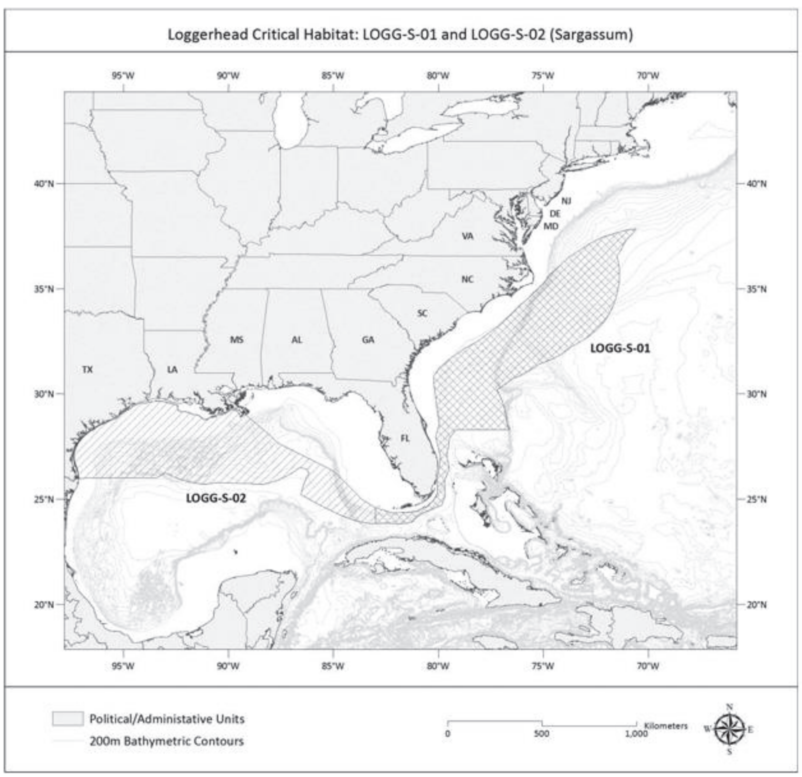

Figure 29. ***Sargassum*** **critical habitat**

Status of Critical Habitat
NMFS is currently unaware of any adverse impacts to the essential features of the designated critical habitat Unit LOGG-S-02 for loggerheads. NMFS is not aware of any actions that have or are currently destroying or adversely modifying *Sargassum* in critical habitat unit LOGG-S-02 since the listing. There have been no projects destroying or adversely modifying (1)

convergence zones, downwelling areas and other locations where there are concentrated components of the *Sargassum* community, (2) the density or concentration of *Sargassum*, or (3) the prey community associated with *Sargassum* habitat.

Threats to Critical Habitat

Potential threats to loggerhead critical habitat include any activities that adversely affect the essential features. Such potential threats include:

<u>Oil Spills</u>
Large scale oil spills can adversely affect the *Sargassum* units of loggerhead critical habitat thereby reducing their ability to provide developmental and foraging habitat for young loggerheads. Surface oils can accumulate in mats of *Sargassum* and affect the prey community that loggerhead turtles rely on. Additionally, oil spill response activities such as the use of dispersants, in-situ burning, containment booms, and skimmer operations could further affect the essential features of this habitat, by both affecting prey and modifying the concentration of the algal mats.

The DWH oil spill in 2010 is known to have had a detrimental impact on *Sargassum* and the *Sargassum* community that provide essential habitat functions for loggerhead sea turtles. Heavy oil (greater than 5 percent coverage) affected 23 percent of the *Sargassum* (873 to 1,749 square kilometers) in the northern Gulf of Mexico, resulting in a range of lost *Sargassum* area from forgone growth (loss of potential growth for that growing season from *Sargassum* killed by oil) between 4,524 and 9,392 square kilometers. The total combined loss of *Sargassum* from direct loss and foregone growth loss for that year's crop may have been as high as 11,100 square kilometers (DWH Trustees 2015a). Areas of *Sargassum* that experienced lighter oiling may still have been negatively impacted, reducing the habitat function that would normally be provided. The lethal and sublethal effects to *Sargassum*-dependent species, including sea turtles, was not determined, but detrimental impacts are expected to have occurred due to the loss of habitat functions that would have been provided by that area of habitat.

4 ENVIRONMENTAL BASELINE

By regulation, the environmental baseline for an Opinion refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to the listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are also 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, and areas of designated critical habitat that occur in an action area, that will be exposed to effects from the action under consultation. This is important because, in some states or life history stages, or areas of their ranges, listed individuals or critical habitat features 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.

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and the ecosystem, within the action area. The environmental baseline is a “snapshot” of a species’ health at a specified point in time. It does not include the effects of the action under review in this consultation.

4.1 Status of Species and Critical Habitat within the Action Area

The status of species and critical habitat in the action area, as well as the threats to these species and critical habitat, is supported by the species accounts in Section 3 (Status of Species and Critical Habitat).

As stated in Section 2.2 (Action Area), the proposed SPOT DWP and TGL DWP would be located on the continental shelf of the GoM, in Federal waters, in Galveston Area OCS lease blocks approximately 27.2 to 30.8 n mi off the coast of Brazoria County, Texas.

4.1.1 Sperm Whales

Sperm whale groups have been observed throughout the Gulf of Mexico from the upper continental slope near the 100-meter isobath to the seaward extent of the United States Exclusive Economic Zone (EEZ) and beyond (Baumgartner et al. 2001; Burks et al. 2001; Roden and Mullin 2000). Aggregations of sperm whales are commonly found in waters over the shelf edge in the vicinity of the Mississippi River Delta in waters that are 1,641-6,562 ft (500-2,000 m) in depth (Davis et al. 2000; Davis and Fargion 1996). They are also often concentrated along the continental slope in or near cyclones and zones of confluence between cyclones and anticyclones (Davis et al. 2000). Sperm whales appear to be concentrated in at least two geographic regions of the Northern Gulf of Mexico: an area off the Dry Tortugas and offshore of the Mississippi River delta (Maze-Foley and Mullin 2006). Davis et al. (2000) noted the presence of a resident, breeding population of endangered sperm whales within 50 km of the Mississippi River Delta and suggested that this area may be essential habitat for sperm whales. Consistent sightings and satellite tracking results indicate that sperm whales occupy the northern Gulf of Mexico throughout all seasons (Davis et al. 2000; Davis and Fargion 1996; Jefferson and Schiro 1997; Jochens et al. 2008; Mullin et al. 1994; Sparks et al. 1995). For management purposes, sperm whales in the Gulf of Mexico are considered a separate stock from those in the Atlantic and Caribbean (Engelhaupt et al. 2009; Gero and Whitehead 2007; Jaquet 2006; Jochens et al. 2008). The best abundance estimate available for sperm whales in the northern Gulf of Mexico is 763 individuals (Waring 2016).

Research on the genetic stock structure of Gulf of Mexico sperm whales, gender composition, and kinship patterns indicate a distinct matrilineal population structure of sperm whales in the Gulf of Mexico (Jochens et al. 2008). In this study, 89 individuals (including satellite-tagged, D-tag tagged, opportunistic, and stranded whales) were genotyped using both mtDNA and microsatellite techniques and gender determined using molecular sexing techniques. The majority of whales sampled from groups throughout the north-central Gulf of Mexico fit the classic 'mixed' group scenario, comprised of females and subadults of both sexes. A comparative analysis of matrilineal mtDNA and biparentally inherited nuclear genetic markers has begun to show population structure for these female lineages. Only four mtDNA haplotypes were found in the northern Gulf, with two that are unique on a global scale to this geographic area (Jochens et al. 2008).

BOEM's Sperm Whale Seismic Study provides further conclusions about sperm whales in the northern Gulf of Mexico (Jochens et al. 2008). This study concluded:

1. The data supports the conservation of sperm whales in the northern Gulf of Mexico as a discrete stock.
2. Sperm whales are present year-round in the Gulf, with females generally having significant site fidelity and males and females exhibiting significant differences in habitat use.
3. The sperm whale population off the Mississippi River Delta likely has a core size of about 140 individuals.
4. Gulf of Mexico sperm whales seem to be smaller in individual size than sperm whales in some other oceans.
5. Some groups of sperm whales in the Gulf were mixed-sex groups of females/immatures and others were groups of bachelor males. The typical group size for mixed groups was ten individuals, which is smaller than group sizes in some other oceans.
6. The typical diving and underwater behaviors of the Gulf's sperm whales are similar to those of animals in other oceans.
7. The typical feeding and foraging behaviors of the Gulf's sperm whales are similar to those of animals in other oceans, although differences in defecation rates suggest possible differences in feeding success.
8. In the otherwise oligotrophic (low productivity) Gulf of Mexico, the eddy field contributes to development of regions of locally high surface productivity that in turn may create conditions favorable for trophic cascade of surface production to the depths where Gulf sperm whales dive to forage.

4.1.2 Sea Turtles

The five species of sea turtles that occur in the action area, green (NA and SA DPSs), hawksbill, Kemp's ridley, leatherback, and loggerhead (NWA DPS), are all highly migratory. Therefore, the statuses of the five species (or the DPS) of sea turtles in the action area, as well as the threats to these species, are best reflected in their range-wide statuses and supported by the species accounts in Section 3.2 (Status of Species). Due to their migratory behavior, loggerheads from other recovery units may be present in the action area. However, the nesting beaches for the Northern Gulf of Mexico Recovery Unit of loggerheads, defined as loggerheads originating from nesting beaches from Franklin County on the northwest coast of Florida through Texas, occur in closer proximity to the action area than other recovery units. Although all recovery units of the DPS may be present, individuals of the Northern Gulf of Mexico Recovery Unit would be susceptible to affects during certain times of year (e.g., impacts to nesting activities and the offshore migration of hatchlings from nesting beaches) that other recovery units would not be susceptible to. The great majority of Kemp's ridley sea turtles nest on Gulf of Mexico beaches, which could leave them more vulnerable to threats within the Gulf than the other sea turtles that have nesting beaches in more widespread locations.

4.1.3 Oceanic Whitetip Shark

There were 56 records of oceanic whitetips in the Gulf of Mexico from 1975-1995 on commercial longline vessels as part of the Southeast Fisheries Science Center (SEFSC) Pelagic Longline Observer Program (Kohler et al., 1998). All records are in areas deeper than 200 meters, the majority of which were mature sized individuals near the 2000 meter bathymetry line within federal waters of the Gulf of Mexico to the EEZ.

According to the status review, one oceanic whitetip shark was tagged in the Gulf of Mexico in the NMFS Co-operative Shark Tagging Program from 1962-1993. We do not have any recent records for this species in the Gulf of Mexico, but there is not much monitoring effort in the region they have previously been caught. In four pelagic longline surveys conducted in 2011 and 2012, no oceanic whitetip sharks were caught (B. Hueter, Mote Marine Laboratory, pers. comm. October 5, 2017). Information in the status review suggests there was an 88 percent decline of oceanic whitetip sharks in the Gulf of Mexico since the 1950's (Young et al. 2016a).

Given the large size of the action area and the wide range of the oceanic whitetip shark, oceanic whitetip sharks could occur throughout the action area. Therefore, the status of oceanic whitetips sharks in the action area, as well as the threats to this species, is supported by the species accounts in Section 3.2 (Status of Species).

4.1.4 Giant Manta Ray

The giant manta ray in the Gulf of Mexico is not common, however there is a known small population at the FGBNMS of more than 70 individuals (Miller and Klimovich 2017). It is thought that FGBNMS in the Gulf of Mexico are important nursery areas for juvenile manta rays (Stewart et al., in press). Some individual mantas may be observed in coastal areas occasionally, though we consider such sightings rare, and according to the status review, the giant manta ray (i.e., the larger of the two manta species) may be more oceanic than the reef manta ray.

Given the large size of the action area and the wide range of the giant manta ray, individuals could occur throughout the action area. Therefore, the status of giant manta rays in the action area, as well as the threats to this species, is supported by the species accounts in Section 3.2.7 (Status of Species).

4.1.5 Loggerhead Critical Habitat Unit LOGG-S-02

The status of Loggerhead Critical Habitat Unit LOGG-S-02 in the action area, as well as the threats to this critical habitat, are best reflected in its range-wide status in Section 3.4.

4.2 Factors Affecting Species and Critical Habitat within the Action Area

<u>4.2.1 ESA Section 7 Consultations</u>

Coastal Migratory Pelagics Fishery

The Coastal Migratory Pelagics (CMP) Fisheries Management Plan (FMP) was approved in 1982 and implemented by regulations effective in February of 1983. Managed species include king mackerel, Spanish mackerel, and cobia. The CMP FMP manages these species in federal waters in the Gulf of Mexico and in the Atlantic from Florida to New York. Spanish mackerel occur to depths of 75 m, cobia to depths of 125 m, and king mackerel to depths of 200 m. Consequently, fishing for CMP species typically occurs in waters less than 45 m but may occur in depths as great as 200 m. Fishing for CMP species in the Gulf of Mexico is primarily conducted by hook-and-line, cast nets, and run-around and sink gillnets. Drift gillnets targeting CMP species have been prohibited since 1990, and many additional restrictions on gillnets targeting CMP were implemented in April 2000 via Amendment 9 to the CMP FMP.

Only the gillnet component of the authorized CMP fishery is known to adversely affect sea turtles. While sea turtles are typically vulnerable to capture on hooks, the hook-and-line gear used by both commercial and recreational fishers to target CMP species is limited to trolled or, to a much lesser degree (e.g., historically ~2% by landings for king mackerel), jigged handline, bandit, and rod-and-reel gear, i.e., techniques that are extremely unlikely to affect sea turtles (NMFS 2015).

A June 18, 2015 Opinion, as amended via a November 18, 2017 memorandum and attachment, comprises the most recent completed Section 7 consultation on the operation of the CMP fishery in the Gulf of Mexico and South Atlantic. The 2015 Opinion, as amended, concluded that the proposed action is likely to adversely affect but is not likely to jeopardize the continued existence of all of the listed sea turtle species in the Gulf of Mexico (i.e., green North Atlantic and South Atlantic DPS, hawksbill, Kemp's ridley, leatherback, and loggerhead NWA DPS).

On January 22, 2018, NMFS listed the giant manta ray as threatened under the ESA. On January 30, 2018, NMFS listed the oceanic whitetip shark as threatened under the ESA. NMFS is currently in the process of amending the 2015 Opinion to evaluate the effects of the proposed action’s activities on the oceanic whitetip shark, giant manta ray, and Rice’s whale. Information suggests that oceanic whitetip sharks and giant manta rays may be taken in the CMP fisheries and adversely affected.

Pelagic Longline Fishery for Atlantic Highly Migratory Species species
On May 15, 2020 NMFS issued an Opinion on the operation of the pelagic longline fishery for Atlantic Highly Migratory Species (HMS) fisheries as carried out under the 2006 Consolidated Atlantic HMS Fishery Management Plan (2006 Atlantic Consolidated HMS FMP), as amended. This fishery primarily targets swordfish, yellowfin tuna, and bigeye tuna with secondary target species including dolphin, albacore tuna, and certain species of sharks. The 2020 Opinion determined that the pelagic longline (PLL) fishery in the Gulf of Mexico has in the past, and will continue in the future, to cause direct injury and mortality of sperm whales, giant manta, oceanic whitetip shark, and all species of sea turtles that occur in the Gulf (i.e., green North Atlantic and South Atlantic DPS, hawksbill, Kemp's ridley, leatherback, and loggerhead NWA DPS), through hooking and entanglement of these species in the longline fishing gear deployed in this fishery. In analyzing these expected impacts, the Opinion concluded that the continuing execution of this fishery is likely to adversely affect sperm whales, giant manta, oceanic whitetip shark, and sea turtles, but is not likely to jeopardize the continued existence of any of these species.

Atlantic HMS Fisheries (Excluding Pelagic Longline)
On January 10, 2020, NMFS issued an Opinion on the operation of Atlantic HMS fisheries (excluding the pelagic longline fishery) as carried out under the 2006 Consolidated Atlantic HMS FMP, as amended. The non-PLL Atlantic HMS fisheries use a number of gear types that are known to interact with giant manta, oceanic whitetip shark, and sea turtles, including gillnets, bottom longlines, and vertical lines. These fisheries have been in operation for an extended period of time, and have affected and are part of the environmental baseline for giant manta, oceanic whitetip shark, and sea turtles in the action area for this consultation. The 2020 Opinion concluded that the proposed action is likely to adversely affect, but is not likely to jeopardize the continued existence of giant manta, oceanic whitetip shark, and all of the listed sea turtle species in the Gulf of Mexico (i.e., green North Atlantic and South Atlantic DPS, hawksbill, Kemp's ridley, leatherback, and loggerhead NWA DPS).

Gulf of Mexico Reef Fish Fishery
The Gulf of Mexico reef fish fishery uses 2 basic types of gear: spear or powerhead, and hook-and-line gear. Hook-and-line gear used in the fishery includes both commercial bottom longline and commercial and recreational vertical line (e.g., handline, bandit gear, rod-and-reel).

Prior to 2008, the reef fish fishery was believed to have relatively moderate levels of sea turtle bycatch attributed to the hook-and-line component of the fishery (i.e., approximately 107 captures and 41 mortalities annually, among green, hawksbill, loggerhead, Kemp's ridley and leatherback sea turtles combined, for the entire fishery) (NMFS 2005). In 2008, SEFSC observer programs and subsequent analyses indicated that the overall amount of incidental take for ESA-listed sea turtles specified in the Incidental Take Statement (ITS) of the 2005 Opinion on the reef fish fishery had been severely exceeded by the bottom longline component of the fishery: approximately 974 captures and at least 325 mortalities were estimated for the one-year period from July 2006-2007 for all of the above-listed sea turtle species combined.

In response, NMFS published an Emergency Rule essentially closing the bottom longline sector of the reef fish fishery in the eastern Gulf of Mexico for 6 months pending the implementation of a long-term management strategy. The Gulf of Mexico Fishery Management Council developed a long-term management strategy via a new amendment (Amendment 31 to the Reef Fish FMP). The amendment included: (1) a prohibition on the use of bottom longline gear in the Gulf of Mexico reef fish fishery, shoreward of a line approximating the 35-fathom contour east of Cape San Blas, Florida, from June through August and; (2) a reduction in the number of bottom longline vessels operating in the fishery via an endorsement program and a restriction on the total number of hooks that may be possessed onboard each Gulf of Mexico reef fish bottom longline vessel to 1,000, only 750 of which may be rigged for fishing.

On October 13, 2009, NMFS Southeast Regional Office (SERO) completed an Opinion that analyzed the expected effects of the operation of the Gulf of Mexico reef fish fishery under the changes proposed in Amendment 31 (NMFS-SEFSC 2009b). The Opinion concluded that ESA-listed sea turtle takes would be substantially reduced compared to the fishery as it was previously prosecuted, and that operation of the fishery would not jeopardize the continued existence of green, hawksbill, loggerhead, Kemp's ridley or leatherback sea turtles. Amendment 31 was implemented on May 26, 2010. In August 2011, consultation was reinitiated to address the DWH oil spill event and potential changes to the environmental baseline. Reinitiation of consultation was not related to any material change in the fishery itself. The resulting September 30, 2011, Opinion concluded the operation of the Gulf of Mexico reef fish fishery is not likely to jeopardize the continued existence of green, hawksbill, Kemp's ridley, leatherback, or the Northwest Atlantic DPS of loggerhead sea turtles (NMFS 2011).

In 2018, ESA Section 7 consultation was reinitiated for this fishery to address new and updated listings for green sea turtles (listing 8 new green sea turtle DPSs as threatened and 3 new green sea turtle DPSs as endangered; the North Atlantic and South Atlantic DPSs were the only two affected by this fishery), Nassau grouper, oceanic whitetip shark, and giant manta ray. This consultation is still under way at this time, with an estimated completion date set for the end of 2022.

Southeastern Shrimp Trawl Fisheries
The Southeast U.S. Shrimp Fishery, which employs both otter trawling and skimmer trawling techniques in the action area. Impacts of this fishery on giant manta, oceanic whitetip shark, and five sea turtle species (i.e., Kemp's ridley, loggerhead, green, leatherback and hawksbill) were recently evaluated through Section 7 consultation (NMFS 2021). This and previous consultations on this fishery resulted in mandatory terms and conditions, and subsequent rulemaking to reduce the impacts of the fishery on sea turtle populations. Examples include mandatory use of turtle excluder devices (TEDs) and other gear and trawl-time restrictions, along with robust monitoring, sampling and ecological studies of the shrimp fishery effects on sea turtles.

The 2021 Opinion on the Southeast U.S. Shrimp Fishery (2021 Shrimp Opinion) determined that fishing activities as considered would not adversely affect oceanic whitetip shark. The Opinion also determined that the proposed fishing activities would adversely affect, but not likely to jeopardize, giant manta and the 5 species of sea turtles.

Federal Dredging Activities
Marine dredging for construction and maintenance of federal navigation channels and dredging for borrow materials for marsh creation/restoration are common within the action area. Hopper dredges are capable of moving relatively quickly compared to sea turtle swimming speed and can thus overtake, entrain, and kill sea turtles as the suction draghead(s) of the advancing dredge overtakes the resting or swimming turtle. Entrained sea turtles rarely survive. To reduce impacts to sea turtles, relocation trawling may be utilized to capture and relocate sea turtles. In relocation trawling, a boat equipped with nets precedes the dredge to capture turtles and then releases the animals out of the dredge pathway, thus avoiding lethal take.

Other forms of dredging such as mechanical and hydraulic suction dredging utilize slower moving underwater equipment that is not expected to contact or injure sea turtles. However, these dredging techniques, along with hopper dredging, can cause impacts to aquatic habitat utilized by sea turtles, including: 1) direct removal/burial of prey organisms; 2) turbidity/siltation effects; 3) contaminant re-suspension; and 4) noise/disturbance.

In summary, dredging to maintain navigation channels and dredging/placement of sediments for habitat restoration occurs frequently within the action area.

In 2003, we completed formal consultation on the impacts of USACE's hopper-dredging operations in the Gulf of Mexico (i.e., Gulf of Mexico Regional Biological Opinion [GRBO]). We revised the GRBO in 2007 (NMFS 2007c), in which we concluded that: 1) Gulf of Mexico hopper dredging would adversely affect 4 sea turtle species (i.e., green, hawksbill, Kemp's ridley, and loggerheads), but would not jeopardize their continued existence; and 2) dredging in the Gulf of Mexico would not likely adversely affect leatherback sea turtles.

The GRBO considers maintenance dredging and sand mining operations. We have conducted numerous other informal consultations that analyzed non-hopper dredging projects (e.g., marsh and beach restoration projects) that did not fall partially or entirely under the scope of actions contemplated by the GRBO. All of these consultations have determined that the proposed actions would not likely adversely affect any species of sea turtles or other listed species, or critical habitat of any listed species.

Federal Vessel Activity

Watercraft are the greatest contributors to overall noise in the sea and have the potential to interact with sea turtles though direct impacts by the vessels or their propellers. Sound levels and tones produced are generally related to vessel size and speed. Larger vessels generally emit more sound than smaller vessels, and vessels underway with a full load, or those pushing or towing a load, are noisier than unladen vessels. Vessels operating at high speeds have the potential to strike sea turtles. Potential sources of adverse effects from federal vessel operations in the action area include operations of the Bureau of Ocean Energy Management (BOEM), Federal Energy Regulatory Commission, USCG, NOAA, and USACE. We have conducted Section 7 consultations with all of these agencies, analyzing effects of federal vessel operation in the Gulf of Mexico. Many of these consultations required or confirmed the implementation of conservation measures for vessel operations, designed to avoid or minimize adverse effects to listed species. At the present time, federal vessel operation in the action area continue to present the potential for harassment and injury to ESA-listed species in the action area.

Bluewater Deepwater Oil Port

The Bluewater Deepwater Oil Port project is currently under consultation (SERO-2020-03713). The potential effects of this project on ESA-listed species are under active analysis and have not yet been determined.

4.2.2 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 database[8] by the consulting biologist on January 20, 2022, there were 9 active Section 10(a)(1)(A) scientific research permits applicable to green, Kemp's ridley, loggerhead, and leatherback sea turtles, ESA-listed corals, and ESA-listed elamsobranchs within the State of Texas. These permits allow the capture, handling, sampling, and release of ESA-listed species (all life stages except hatchlings) and range in purpose from reducing bycatch in commercial fisheries to gaining better scientific knowledge. All of the biological opinions issued for these activities concluded that the effects of the actions (issuance of research permits) would not jeopardize any ESA listed species.

4.2.3 State and Private Actions

Recreational fishing as regulated by the State of Texas can affect protected species or their habitats within the action area. Pressure from recreational fishing in and adjacent to the action area is likely to continue. Observations of state recreational fisheries have shown that loggerhead sea turtles are known to bite baited hooks and frequently ingest the hooks. Hooked sea turtles have been reported by the public fishing from boats, piers, and beach, banks, and jetties and from commercial anglers fishing for reef fish and for sharks with both single rigs and bottom longlines (NMFS 2001). Additionally, lost fishing gear such as line cut after snagging on rocks, or discarded hooks and line, can also pose an entanglement threat to sea turtles in the area. A detailed summary of the known impacts of hook-and-line incidental captures to Kemp's ridley and loggerhead sea turtles can be found in the Turtle Expert Working Group (TEWG) reports (1998; 2000).

4.2.4 Marine Debris

The discharge of debris into the marine environment is a continuing threat to the status of species in the action area, regardless of whether the debris is discharged intentionally or accidentally. Marine debris may originate from a variety of sources, though specific origins of debris are difficult to identify. A 1991 report (GESAMP 1990) indicates that up to 80 percent of marine debris is considered land-based and a worldwide review of marine debris identifies plastic as the primary form (Derraik 2002). Debris can originate from a variety of marine industries including fishing, oil and gas, and shipping. Many of the plastics discharged to the sea can withstand years of saltwater exposure without disintegrating or dissolving. Further, floating materials have been shown to concentrate in ocean gyres and convergence zones where *Sargassum* and consequently juvenile sea turtles are known to occur (Carr 1987).

Marine debris has the potential to impact ESA-listed species through ingestion or entanglement (Gregory 2009). Both of these effects could result in reduced feeding, reduced reproductive success, and potential injury, infection, or death. Sperm whale ingestion of marine debris is a concern, particularly because the suspected feeding behavior of these whales includes cruising along the bottom with their mouths open (Walker and Coe 1990). All sea turtles are susceptible to ingesting marine debris, though leatherbacks show a marked tendency to ingest plastic which they misidentify as jellyfish, a primary food source (Balazs 1985b). Ingested debris may block the digestive tract or remain in the stomach for extended periods, thereby reducing the feeding drive, causing ulcerations and injury to the stomach lining, or perhaps even providing a source of

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

toxic chemicals (Laist 1987; Laist 1997). Weakened animals are then more susceptible to predators and disease and are also less fit to migrate, breed, or, in the case of turtles, nest successfully (Katsanevakis 2008; McCauley and Bjorndal 1999).

Pollution from a variety of sources including atmospheric loading of pollutants such as PCBs, stormwater from coastal or river communities, and discharges from ships and industries may affect sea turtles, sperm whales, giant manta ray and oceanic whitetip sharks in the action area. Sources of marine pollution are often difficult to attribute to specific federal, state, local or private actions.

There are studies on organic contaminants and trace metal accumulation in green, leatherback, and loggerhead sea turtles (Aguirre et al. 1994; Caurant et al. 1999; Corsolini et al. 2000). McKenzie et al. (1999) measured concentrations of chlorobiphenyls and organochlorine pesticides in sea turtles tissues collected from the Mediterranean (Cyprus, Greece) and European Atlantic waters (Scotland) between 1994 and 1996. Omnivorous loggerhead turtles had the highest organochlorine contaminant concentrations in all the tissues sampled, including those from green and leatherback turtles (Storelli et al. 2008). It is thought that dietary preferences were likely to be the main differentiating factor among species. Decreasing lipid contaminant burdens with sea turtle size were observed in green turtles, most likely attributable to a change in diet with age. (Sakai et al. 1995) documented the presence of metal residues occurring in loggerhead sea turtle organs and eggs. Storelli et al. (1998) analyzed tissues from 12 loggerhead sea turtles stranded along the Adriatic Sea (Italy) and found that characteristically, mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991). No information on detrimental threshold concentrations is available and little is known about the consequences of exposure of organochlorine compounds to sea turtles. Research is needed on the short- and long-term health and fecundity effects of chlorobiphenyl, organochlorine, and heavy metal accumulation in sea turtles.

Sperm whales may be adversely affected by marine pollution originating from federal, state, or private activities, though little is known regarding the specific pollutants or the effects pollutants may have on individuals. Further, we are unaware of the possible long-term and trans-generational effects of exposure to pollutants. We do not know if high levels of heavy metals, PCBs, and organochlorines found in prey species accumulate with age or are transferred through nursing. Nevertheless, the accumulation of stable pollutants such as heavy metals, PCBs, chlorinated pesticides [DDT, DDE, etc.], and polycyclic aromatic hydrocarbons [PAHs]) is of concern.

The development of marinas and docks in inshore waters can negatively impact nearshore habitats. 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 of the action area, the species of turtles analyzed in this biological opinion travel between nearshore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles. Fuel oil spills could affect animals directly or indirectly through the food chain. Fuel spills involving fishing vessels are common events. However, these spills typically involve small amounts of material. Larger oil spills may result from accidents, although these events would be rare. No direct adverse effects on listed species resulting from fishing vessel fuel spills have been documented.

4.2.5 Acoustic Impacts

NMFS has established criteria to predict varying levels of responses of marine species to anthropogenic sound, based upon hearing injury and behavioral responses (https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-acoustic-technical-guidance). Responses to sound exposure may include lethal or nonlethal injury, temporary hearing impairment, behavioral harassment and stress, or no apparent response. Contributions to ambient sound levels include vessels, geophysical exploration, and the construction, operation, and decommissioning of offshore structures. It is expected that the policy on managing anthropogenic sound in the oceans will provide guidance for programs such as incidental harassment permits under the Marine Mammal Protection Act and permits for research involving sound-producing activities. NOAA is working cooperatively with the ship-building industry to find technologically-based solutions to reduce the amount of sound produced by commercial vessels. Through ESA consultation with NMFS, the Bureau of Ocean Energy Management (BOEM) and Bureau of Safety and Environmental Enforcement (BSEE) have implemented and periodically revised Gulf of Mexico-wide measures, such as 2016 Notice to Lessees and Operators regarding the implementation of seismic survey mitigation measures and protected species observer program (BOEM NTL 2016-G02) (https://www.boem.gov/sites/default/files/documents/oil-gas-energy/BOEM-NTL-No-2016-G02.pdf), to reduce the risk of harassment to sperm whales from sound produced by geological and geophysical surveying activities and the explosive removal of offshore structures.

NOAA has implemented the CetSound Ocean Sound Strategy (http://cetsound.noaa.gov/) that provides for a better understanding of man-made sound impacts on cetacean species. CetSound produced modeled ambient sound maps for several sound source types in the Gulf of Mexico. Annual average ambient sound sums of the modeled source types including seismic airgun surveys at different frequencies and depths is displayed in Figure 30. Other modeled events that can be viewed on the CetSound website for the Gulf of Mexico include annual average ambient sound for only seismic airguns surveys, summed sound sources without airguns, and explosive severance of an oil platform during decommissioning.

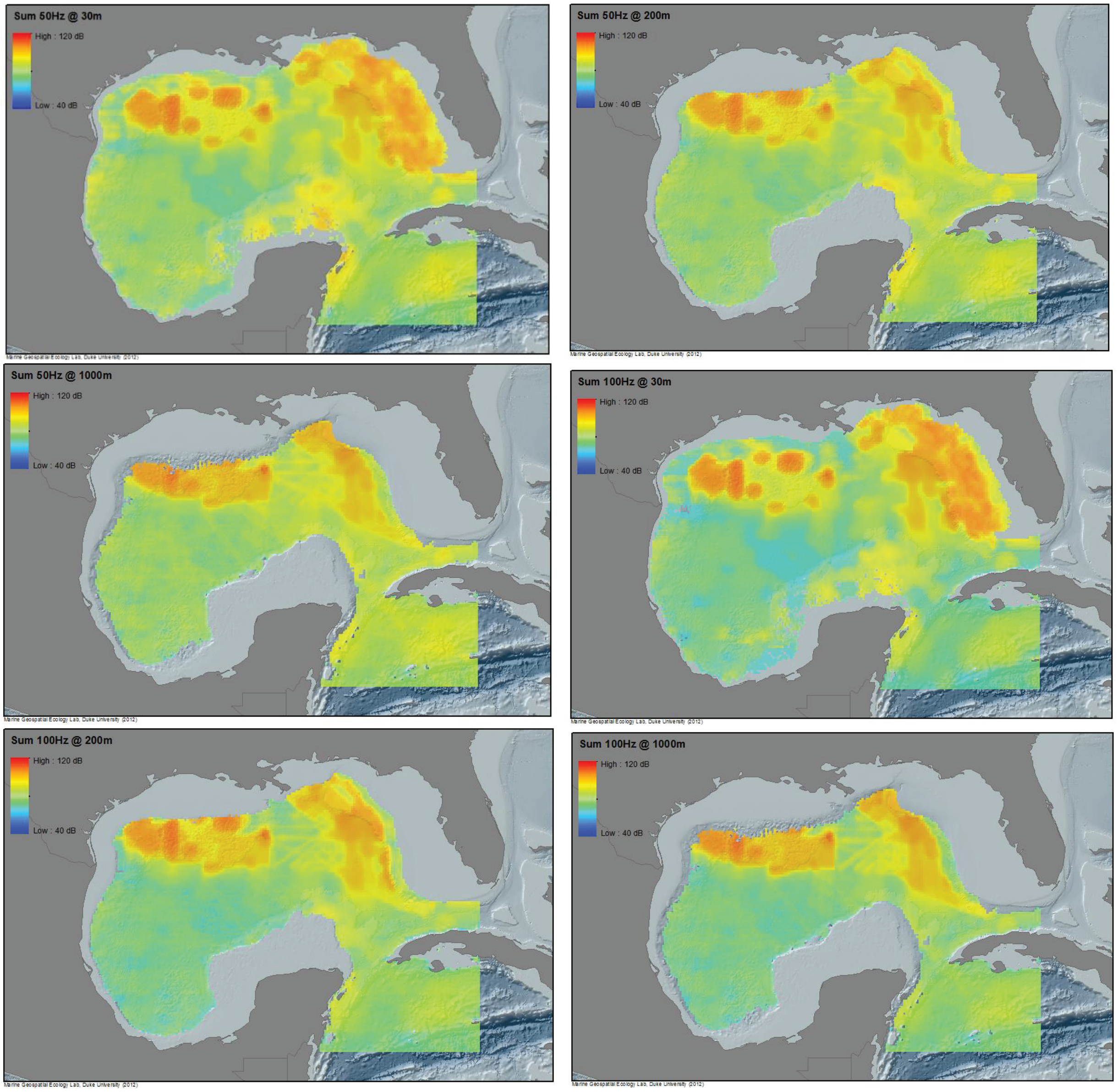

Figure 30. Predicted average contribution to ambient sound from modeled sound sources including seismic airgun surveys at different depths for 50 Hz and 100 Hz. Source: Marine Geospatial Ecology Lab, Duke University (2012) as published on CetSound website.

The Gulf of Mexico soundscape is being studied over the long-term by NOAA's Sound Reference Station Network (https://www.pmel.noaa.gov/acoustics/noaanps-ocean-noise-reference-station-network). This network uses static passive acoustic monitoring (PAM) hydrophone (sound recorder) units to monitor trends and changes in the ambient sound field in U.S. federal waters. In addition to this network, there have been several other hydrophone units in the northern Gulf of Mexico (Figure 31). A study by Wiggins et al. (2016) placed two high-frequency acoustic recording packages (HARPs) in 100 m to 250 m water depths and three HARPs in approximately 1,000 m water depth to compare low-frequency sound pressure spectrum levels over three years.

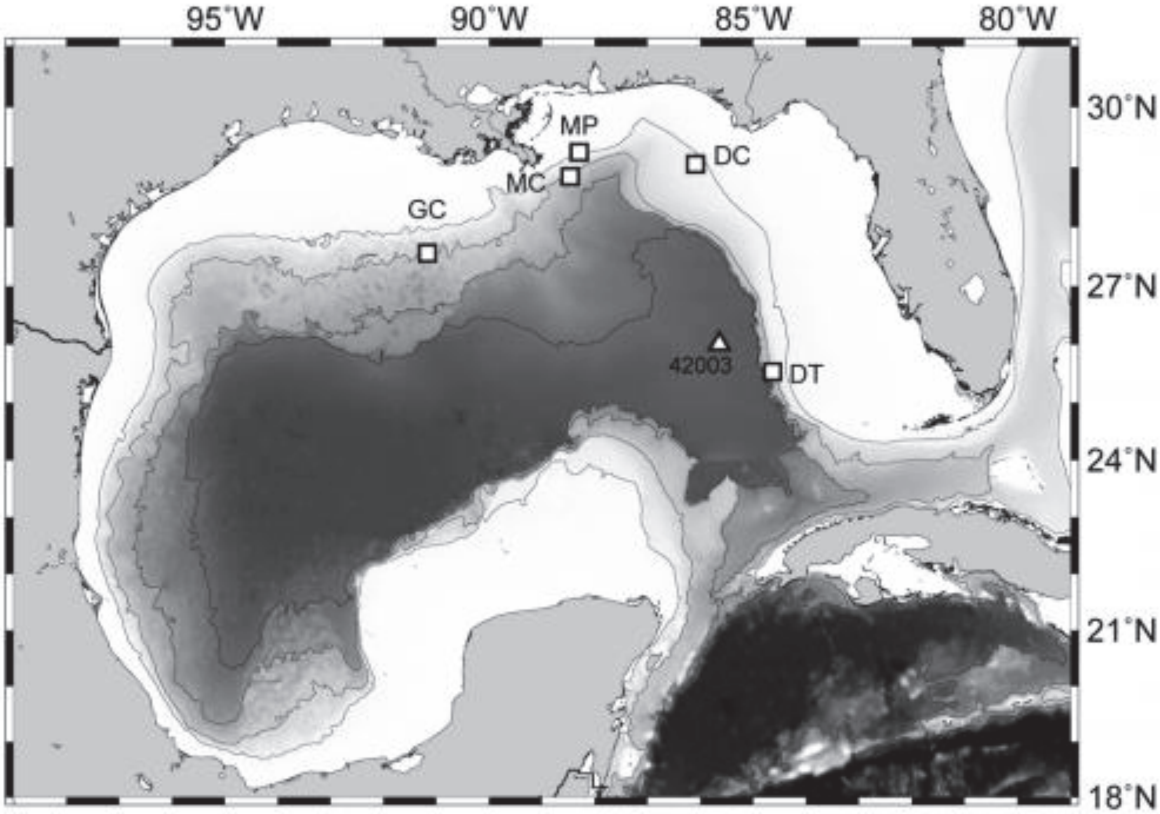

Figure 31. Five HARP locations, which collected data over several months during 2010-2013, are displayed as squares notated with site codes [GC: Green Canyon; MC: Mississippi Canyon; MP: Main Pass; DC: De Soto Canyon; and DT: Dry Tortugas]. The triangle is a NOAA weather bouy station used to measure wind speeds. Figure from (Wiggins et al. 2016).

The (Wiggins et al. 2016) study concluded:

1. Deepwater sites (Green Canyon, Mississippi Canyon and Dry Tortugas) had the highest sound pressure levels below 100 Hz and they reported some of the highest measured averages over long periods.
2. Gulf of Mexico ship traffic, especially bulk carriers, is of the highest ranked in U.S. ports; however, seismic airgun pulses are the dominant source of low frequency, high sound levels in the deep water.
3. When a Hurricane swept through in August 2012, the sound pressure levels being recorded dropped by over 10 to 81 dB re $1\mu Pa^2$ at 40 Hz, likely due to cessation of shipping and seismic operations.
4. Shallow sites (Main Pass, De Soto Canyon) differed from each other and from the deepwater sites and generally were quieter than the deeper sites. This was attributed to proximity far from anthropogenic activity.

Sound is a stressor that is produced by many activities discussed in the baseline sections below.

4.4.6 Oil spills

There are many oil spills each year in the GoM due to this region's very large number of subsea wells, offshore production structures, pipelines, vessels, and other infrastructure supporting oil and gas activities.

The action area are within the BOEM GoM Western Planning Area (WPA). Table 10 and Table 11 summarize BOEM's estimated probability of spills over 1,000 barrels (bbl) and very large spills over 10,000 bbl resulting from pipeline, platform, and tanker accidents over a 50 year period in the WPA. These estimates are specific to oil and gas exploration and development activities, and do not include spills that might result from distribution activities (such as the proposed actions). The highest likelihood of occurrence of spills is from pipelines, followed by platforms, and a lower occurrence from tankers. The high risk from pipelines is due to the fact that nearly all of the oil produced in the GoM is transported to shore via the vast pipeline infrastructure found offshore. Accidental pipeline breaks from underwater landslides, anchoring, storms, and an aging pipeline infrastructure are the causes of most pipeline spills. Several small to medium (from 1,000 to 10,000 bbl), and a few large(over 10,000 bbl) spills from platforms, pipelines, and tankers are predicted to occur in the future.

Table 10. Likelihood of Occurrence for Oil Spills over 1,000 bbl from Platforms, Pipelines, and Tankers in the BOEM GoM Western Planning Area (WPA) (2012-2051) (NMFS, 2020)

Planning Area	Probability of One or More Spills (Percent Chance)			Total Chance
	Platforms	Pipelines	Tankers	
WPA	47-60 percent	89-94 percent	0-17 percent	94-98 percent

Table 11. Likelihood of Occurrence for Oil Spills over 10,000 bbl from Platforms, Pipelines, and Tankers in the BOEM GoM (WPA) (2012-2051) (NMFS, 2020)

Planning Area	Probability of One or More Spills (Percent Chance)			Total Chance
	Platforms	Pipelines	Tankers	
WPA	28-38 percent	36-43 percent	0.0-6.0 percent	54-67 percent

NMFS (2020) estimated the number of oil spills that could be expected in the GoM for a 50 year period of oil and gas exploration and development activity. The number of spills estimated was derived through BOEM's application of the historical rate of spills per volume crude oil handled (1996-2010) (Anderson et al. 2012). Table 12 indicates that numerous small spills are common and will likely continue to occur in the future. Based on BOEM's historical spill data, numerous small spills are expected to occur, but spill frequency decreases as the size of the spill increases.

Table 12. Average Number and Size of Spills projected by BOEM to Occur on the GoM OCS Resulting from Permitted Lease Actions on Leases Awarded through 2027 (NMFS, 2020)

Spill Size Category (bbl)	Median Spill Size (bbl)	Range of the Total Number of Spills over 50 Years	Average Spills over 50 Years
Very Small (1.1-9.9)	3.0	172-344	258
Small (10.0-49.9)	30	52-104	78
Medium (50-499.9)	130	34-68	51
Moderately Large (500.0-999.9)	750	5-10	7.5

Spill Size Category (bbl)	Median Spill Size (bbl)	Range of the Total Number of Spills over 50 Years	Average Spills over 50 Years
Large (1,000-9,999)	2,200	3-7	5
Very Large (≥ 10,000)	100,000	2	2

NMFS (2020) estimated up to two spills greater than 10,000 bbl may occur over the next 50 years of the GoM oil and gas exploration and development. The only oil spill greater than 10,000 bbl to occur within the Gulf of Mexico in the last 20 years was the DWH spill.

DWH Oil Spill

On April 20, 2010, the semi-submersible drilling rig DWH, located approximately 50 miles offshore Louisiana, experienced an explosion and fire. The rig sank and oil and natural gas was released into the Gulf of Mexico. Oil flowed for 86 days, until the well was capped on July 15, 2010. Millions of barrels of oil were released. Additionally, approximately 1.84 million gallons of chemical dispersant was applied both subsurface and on the surface to attempt to break down the oil. Effects of the spill went beyond the footprint that was visually detected through satellite. Berenshtein et al. (2020) used in-situ observations and oil spill transport modeling to examine the full extent of the DWH spill. Figure 32 below displays visible and toxic (brown); invisible and toxic (yellow) and non-toxic (blue) oil concentrations.

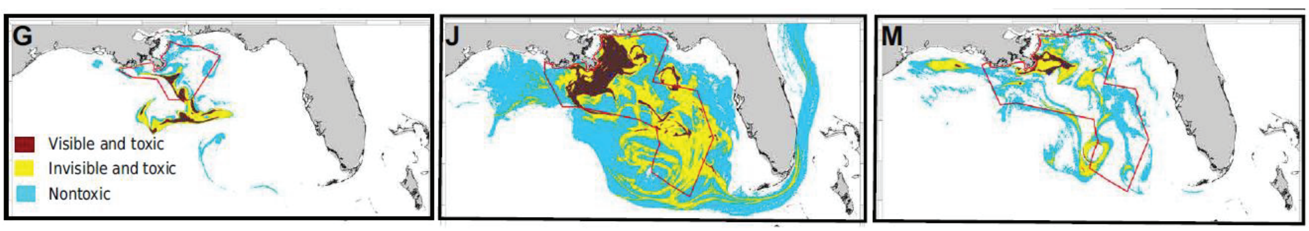

Figure 32. Figure from Berenshtein et al. (2020a) showing spatiotemporal dynamics of the spill for dates showing cumulative oil concentrations in figures G- 15 May 2010; J- 18 June 2010; and M- 2 July 2010

The DWH spill and associated response activities (e.g., skimming, burning, and application of dispersants) resulted in adverse effects on ESA-listed sea turtles, sperm whales, Rice's whales, and Gulf sturgeon. The DWH oil spill extensively oiled vital foraging, migratory, and breeding habitats of ESA-listed sea turtles throughout the northern GoM. *Sargassum* habitats, benthic foraging habitats, surface and water column waters, and sea turtle nesting beaches were all affected by DWH. Sea turtles were exposed to DWH oil in contaminated habitats; breathing oil droplets, oil vapors, and smoke; ingesting oil-contaminated water and prey; and by maternal transfer of oil compounds to developing embryos. Translocation of eggs from the Gulf of Mexico to the Atlantic coast of Florida resulted in the loss of sea turtle hatchlings to the Gulf populations. Other response activities, including vessel strikes and dredging also resulted in turtle deaths.

NMFS (2020) estimated that as many as 7,590 large juvenile and adult sea turtles (Kemp's ridleys, loggerheads, and unidentified hardshelled sea turtles), and up to 158,900 small juvenile sea turtles (Kemp's ridleys, green turtles, loggerheads, hawksbills, and hardshelled sea turtles not identified to species) were killed by the DWH oil spill (Table 13 and Table 14). Small juveniles were affected in the greatest numbers and suffered a higher mortality rate than large sea turtles. Leatherback foraging and migratory habitat was also affected and though impacts to leatherbacks were unquantified, it is likely some died as a result of the DWH spill and spill response (NMFS USFWS 2013; Trustees 2016).

Table 13. Oceanic Juvenile Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley	206,000	35,500	51,000	86,500
Loggerhead	29,800	2,070	8,310	10,400
Green	148,000	15,300	39,800	55,100
Hawksbill	8,560	595	2,390	2,990
Unidentified	9,960	1,310	2,600	3,910
Total	402,320	54,775	104,100	158,900

Table 14. Large Juveniles and Adult Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley age 4+	21,000	1,700	950	2,700
Kemp's ridley age 3	990	380	30	410
Kemp's, all	22,000	2,100	980	3,100
Loggerhead	30,000	2,200	1,400	3,600
Unidentified	5,900	630	260	890
Total	57,900	4,930	2,640	7,590

McDonald et al. (2017c) estimated approximately 402,000 surface-pelagic sea turtles were exposed with 54,800 likely to have been heavily oiled. Additionally, approximately 30 percent of all oceanic turtles affected by DWH and not heavily oiled were estimated to have died from ingestion of oil (Mitchelmore et al. 2017).

With regards to ESA-listed whales, NOAA (2015) determined that 16 percent of the GoM sperm whale population (or about 262 whales) were exposed to DWH oil. Thirty-five percent of those whales (or approximately 92 whales) were likely killed. Nearly half of the population of the GoM Rice's whale was impacted by DWH oil, resulting in an estimated 22 percent maximum decline in population size (Trustees 2016).

Despite natural weathering processes since the DWH spill, oil persists in some habitats and continues to impact exposed resources in the northern GoM (BOEM 2016; Trustees 2016, NMFS, 2020). The true impacts to offshore megafauna populations and their habitats may never be fully quantified (Frasier 2020).

4.4.7 Climate Change

In addition to the information on climate change presented in the Section 3 (Status of the Species) for sea turtles, the discussion below presents further background information on global climate change as well as past and predicted future effects of global climate change we expect within the action area. Also, below is the best available information on predicted effects of climate change in the action area and how listed sea turtles may be affected by those predicted environmental changes. The effects are summarized on the time span of the proposed actions, for which we can realistically analyze impacts, yet are discussed and considered for longer time periods when feasible. Yet, as mentioned previously, the potential effects, and the expected related climate change effects to ESA-listed species and their habitats, are the result of slow and steady shifts or alterations over a long time-period, and forecasting any specific critical threshold that may occur at some point in the future (e.g., several decades) is fraught with uncertainty. As a result, for the purposes of this Opinion we have elected to view the effects of climate change on affected species on a more manageable and predictable 10-year time period due to this reality. While climate change is also relevant to the Cumulative Effects section of this Opinion, we are synthesizing all additional information here rather than include partial discussions in other sections of this Opinion.

In order to evaluate the implications of different climate outcomes and associated impacts throughout the 21st century, many factors have to be considered. The amount of future greenhouse gas emissions is a key variable. Developments in technology, changes in energy generation and land use, global and regional economic circumstances, and population growth must also be considered.

A set of four scenarios was developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections are employed consistently across the various branches of climate science. The scenarios are referred to as representative concentration pathways (RCPs), which capture a range of potential greenhouse gas emissions pathways and associated atmospheric concentration levels through 2100 (IPCC 2014). The RCP scenarios drive climate model projections for temperature, precipitation, sea level, and other variables: RCP2.6 is a stringent mitigation scenario; RCP2.5 and RCP6.0 are intermediate scenarios; and RCP8.5 is a scenario with no mitigation or reduction in the use of fossil fuels. The IPCC future global climate predictions (2014 and 2018) and national and regional climate predictions included in the Fourth National Climate Assessment for U.S. states and territories (2018) use the RCP scenarios.

The increase of global mean surface temperature change by 2100 is projected to be 0.3 to 1.7°C under RCP 2.6, 1.1 to 2.6°C under RCP 4.5, 1.4 to 3.1°C under RCP 6.0, and 2.6 to 4.8°C under RCP8.5 with the Arctic region warming more rapidly than the global mean under all scenarios (IPCC 2014). The Paris Agreement aims to limit the future rise in global average temperature to 2°C, but the observed acceleration in carbon emissions over the last 15 to 20 years, even with a lower trend in 2016, has been consistent with higher future scenarios such as RCP8.5 (Hayhoe et al. 2018).

The globally-averaged combined land and ocean surface temperature data, as calculated by a linear trend, show a warming of approximately 1.0°C from 1901 through 2016 (Hayhoe et al. 2018). The IPCC Special Report on the Impacts of Global Warming noted that human-induced warming reached temperatures between 0.8 and 1.2°C above pre-industrial levels in 2017, likely increasing between 0.1 and 0.3°C per decade. Warming greater than the global average has already been experienced in many regions and seasons, with most land regions experiencing greater warming than over the ocean (Allen et al. 2018). Annual average temperatures have increased by 1.8°C across the contiguous U.S. since the beginning of the 20th century with Alaska warming faster than any other state and twice as fast as the global average since the mid-20th century (Jay et al. 2018). Global warming has led to more frequent heatwaves in most land regions and an increase in the frequency and duration of marine heatwaves (Allen et al. 2018). Average global warming up to 1.5°C as compared to pre-industrial levels is expected to lead to regional changes in extreme temperatures, and increases in the frequency and intensity of precipitation and drought (Allen et al. 2018).

Several of the most important threats contributing to the extinction risk of ESA-listed species, particularly those with a calcium carbonate skeleton such as corals and mollusks as well as species for which these animals serve as prey or habitat, are related to global climate change. The main concerns regarding impacts of global climate change on coral reefs and other calcium carbonate habitats generally, and on ESA-listed corals and mollusks in particular, are the magnitude and the rapid pace of change in greenhouse gas concentrations (e.g., carbon dioxide and methane) and atmospheric warming since the Industrial Revolution in the mid-19th century. These changes are increasing the warming of the global climate system and altering the carbonate chemistry of the ocean [ocean acidification; (IPCC 2014)]. As carbon dioxide concentrations increase in the atmosphere, more carbon dioxide is absorbed by the oceans, causing lower pH and reduced availability of calcium carbonate. Because of the increase in carbon dioxide and other greenhouse gases in the atmosphere since the Industrial Revolution, ocean acidification has already occurred throughout the world's oceans, including in the Caribbean, and is predicted to increase considerably between now and 2100 (IPCC 2014).

According to the best available information, the Atlantic Ocean, including the Gulf of Mexico, appears to be warming faster than all other ocean basins except perhaps the southern oceans (Cheng et al. 2017). Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Since the early 1980s, the annual minimum sea ice extent (observed in September each year) in the Arctic Ocean has decreased at a rate of 11 to 16 percent per decade (Jay et al. 2018). Further, ocean acidity has increased by 26 percent since the beginning of the industrial era. A study by (Polyakov et al. 2009) suggests that the North Atlantic Ocean, including the Gulf of Mexico, overall has been experiencing a general warming trend over the last 80 years of 0.031±0.0006 degrees Celsius per decade in the upper 2,000 meters (6,561.7 feet) of the ocean. Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Since the early 1980s, the annual minimum sea ice extent (observed in September each year) in the Arctic Ocean has decreased at a rate of 11 to 16 percent per decade (Jay et al. 2018). Further, ocean acidity has increased by 26 percent since the beginning of the industrial era (IPCC 2014) and this rise has been linked to climate change. Climate change is also expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, tropical storms, heat waves, and droughts (IPCC 2014). Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure (MacLeod et al. 2005; Robinson et al. 2005). Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure (Evans and Bjørge 2013; IPCC 2014; Kintisch 2006; Learmonth et al. 2006; MacLeod et al. 2005; McMahon and Hays 2006; Robinson et al. 2005).

Though predicting the precise consequences of climate change on highly mobile marine species is difficult (Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring. For example, 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 to 35°C (Ackerman 1997). Increases in global temperature could skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007aa; NMFS and USFWS 2007fb; NMFS and USFWS 2013aa; NMFS and USFWS 2013bb; NMFS and USFWS 2015). These impacts will be exacerbated by sea level rise. The loss of habitat because 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. 2006b).

Changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), ultimately affecting primary foraging areas of ESA-listed species including marine mammals, sea turtles, and fish. Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. They predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback turtles were predicted to gain core habitat area, whereas loggerhead turtles and blue whales were predicted to experience losses in available core habitat. McMahon and Hays (2006) predicted increased ocean temperatures will expand the distribution of leatherback turtles into more northern latitudes. The authors noted this is already occurring in the Atlantic Ocean. (MacLeod 2009) estimated, based upon expected shifts in water temperature, 88 percent of cetaceans will be affected by climate change, with 47 percent predicted to experience unfavorable conditions (e.g., range contraction). Willis-Norton et al. (2015) acknowledged there will be both habitat loss and gain, but overall climate change could result in a 15 percent loss of core pelagic habitat for leatherback turtles in the eastern South Pacific Ocean.

Similarly, climate-related changes in important prey species populations are likely to affect predator populations. For example, blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et al. 1999; Payne et al. 1986; Payne et al. 1990). Pecl and Jackson (2008) predicted climate change will likely result in squid that hatch out smaller and earlier, undergo faster growth over shorter life-spans, and mature younger at a smaller size. This could have negative consequences for species such as sperm whales, whose diets can be dominated by cephalopods. For ESA-listed species that undergo long migrations, if either prey availability or habitat suitability is disrupted by changing ocean temperatures regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott 2009).

This review provides some examples of impacts to ESA-listed species and their habitats within the action area that may occur as the result of climate change. While it is difficult to accurately predict the consequences of climate change to a particular species or habitat, a range of consequences are expected that are likely to change the status of the species and the condition of their habitats.

4.4.8 Conservation and Recovery Actions Shaping the Environmental Baseline

Sperm Whale Conservation and Recovery Actions
In December 2010, NMFS published a final recovery plan for sperm whale. NMFS has established a long-term monitoring network for the collection of acoustic data in all federal waters including the GoM. This information will allow managers to better understand and manage potential sound impacts to this whale species.

The Gulf of Mexico Marine Assessment Program for Protected Species (GoMMAPPS) is an ongoing effort between federal partners that is collecting empirical data during aerial and shipboard vessel surveys, satellite tracking of tagged animals, genetic analysis and with the goal to develop updated density models.

BOEM's Environmental Studies Program
BOEM funds research projects specifically to inform policy decisions regarding development of Outer Continental Shelf energy and mineral resources. Some of these studies may benefit marine fauna by providing more information towards understanding those resources. For example, BOEM collects emissions information related to offshore operations and has established a Gulf-wide emission inventory. BOEM is currently conducting a study to perform dispersion and photochemical modelling for the U.S. portion of the GoM to verify effectiveness of existing air quality emissions exemption thresholds and to ensure annual and short-term National Ambient Air Quality Standards are being met (https://opendata.boem.gov/BOEM-ESP-Ongoing-Study-Profiles-2017-FYQ1/BOEM-ESP-GM-14-01.pdf; BOEM 2014).

There are also summaries for current studies on assessing the effects of anthropogenic stressors on marine mammals and discerning behavioral patterns of sea turtles in the GoM. All of the BOEM studies are available through an online system called the Environmental Studies Program Information System found at https://marinecadastre.gov/espis/#/.

Sea Turtles Conservation and Recovery Actions
NMFS has implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for the Atlantic HMS, South Atlantic snapper-grouper, GOM reef fish fisheries, and TED requirements for the Southeast shrimp trawl fishery. In addition to regulations, outreach programs have been established and data on sea turtle interactions with recreational fisheries has been collected through the Marine Recreational Information Program.

Federal Conservation and Recovery Actions

Critical habitat for loggerhead sea turtles was jointly designated by NMFS and USFWS on July 10, 2014 (79 FR 39856)

Reducing Threats from Pelagic Longline and Other Hook-and-Line Fisheries
On July 6, 2004, NMFS published a Final Rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic pelagic longline fishery (69 FR 40734). The management measures include mandatory circle hook and bait requirements, and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality.

NMFS published the Final Rules to implement sea turtle release gear requirements and sea turtle careful release protocols in the GoM reef fish (August 9, 2006; (71 FR 45428) and South Atlantic snapper-grouper fisheries (November 8, 2011; Lopez-Pujol and Ren 2009). These measures require owners and operators of vessels with federal commercial or charter vessel/headboat permits for Gulf reef fish and South Atlantic snapper-grouper to comply with sea turtle (and smalltooth sawfish) release protocols and have on board specific sea turtle release gear.

Revised Use of Turtle Excluder Devices (TEDs) in Trawl Fisheries
NMFS has also implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles in commercial shrimp trawl fisheries. In particular, NMFS has required the use of TEDs in southeast United States shrimp trawls since 1989 and in summer flounder trawls in the mid-Atlantic area (south of Cape Charles, Virginia) since 1992. It has been estimated that TEDs exclude 97 percent of the sea turtles caught in such trawls. These regulations have been refined over the years to ensure that TED effectiveness is maximized through more widespread use, and proper placement, installation, configuration (e.g., width of bar spacing), and floatation. The NMFS continues to work towards development of new, more effective gear specific to fishery needs.

Placement of Fisheries Observers to Monitor Sea Turtle Captures
On August 3, 2007, NMFS published a Final Rule that required selected fishing vessels to carry observers on board to collect data on sea turtle interactions with fishing operations, to evaluate existing measures to reduce sea turtle captures, and to determine whether additional measures to address prohibited sea turtle captures may be necessary (72 FR 43176). This Rule also extended the number of days NMFS observers could be placed aboard vessels, from 30 to 180 days, in response to a determination by the Assistant Administrator that the unauthorized take of sea turtles may be likely to jeopardize their continued existence under existing regulations.

State Conservation and Recovery Actions
Under Section 6 of the ESA, state agencies may voluntarily enter into cooperative research and conservation agreements with NMFS to assist in recovery actions of listed species. NMFS currently has an agreement with all states along the GoM. Prior to issuance of these agreements, the proposals were reviewed for compliance with section 7 of the ESA.

Other Conservation Efforts
<u>Sea Turtle Handling and Resuscitation Techniques</u>

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

Outreach and Education, Sea Turtle Entanglement, and Rehabilitation
There is a Sea Turtle Stranding Network with extensive participant coverage along the Atlantic and GoM coasts that not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded sea turtles.

A Final Rule (70 FR 42508) published on July 25, 2005, allows any agent or employee of NMFS, the USFWS, the USCG, or any other federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. NMFS already affords the same protection to sea turtles listed as threatened under the ESA [50 CFR §223.206(b)].

NMFS has also been active in public outreach efforts to educate fishermen regarding sea turtle handling and resuscitation techniques. As well as making this information widely available to all fishermen, NMFS recently conducted a number of workshops with Atlantic HMS pelagic longline fishers to discuss bycatch issues including protected species, and to educate them regarding handling and release guidelines. NMFS intends to continue these outreach efforts and hopes to reach all fishers participating in the Atlantic HMS pelagic longline fishery.

Recovery Plans and Reviews
The second revision to the recovery plan for the loggerhead sea turtle was completed January 11, 2009 (NMFS and USFWS 2009). The recovery plan for the Kemp's ridley sea turtle was published 2011 (NMFS et al. 2011a). Recovery teams comprised of sea turtle experts have been convened and are currently working towards revising these plans based upon the latest and best available information. Five-year status reviews were completed in 2013 for hawksbill and leather back sea turtles, and in 2015 for green, and Kemp's ridley sea turtles. A review of the loggerhead sea turtle's status was conducted in 2009 (Conant et al. 2009a). These reviews were conducted to comply with the ESA mandate for periodic status evaluation of listed species to ensure that their threatened or endangered listing status remains accurate. Both loggerhead and green sea turtles were reclassified under the ESA (76 FR 58868; 80 FR 15271).

5 EFFECTS OF THE ACTION ON SPECIES AND CRITICAL HABITAT

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

In this section of our Opinion, we assess the effects of the actions on listed species that are likely to be adversely affected. The analysis in this section forms the foundation for our jeopardy analysis in Section 7.0. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Data are limited, so we are often forced to make assumptions to overcome the limits in our knowledge. Sometimes, the best available information may include a range of values for a particular aspect under consideration, or different analytical approaches may be applied to the same data set. In those cases, the uncertainty is resolved in favor of the species (House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)). NMFS generally selects the value that would lead to conclusions of higher, rather than lower risk to endangered or threatened species. This approach provides the "benefit of the doubt" to threatened and endangered species.

Threats From a Crude Oil Spill

The only route of effect from the proposed actions that we believe is likely to adversely affect ESA-listed species is the potential for an oil spill related to the operation of the SPOT DWP and the TGL DWP. Based on decades of experience with oil and gas transportation projects in the Gulf of Mexico, we reasonably assume one or more oil spill(s) associated with the proposed actions will occur over the next 30 years. Oil spills associated with the proposed actions could occur for a number of reasons including equipment failure, human error, natural forces such as hurricanes, or a combination of causal factors. Sources of spills include the deepwater port platform and structures, vessels, and pipelines. Oil spills are accidental and unpredictable events, but would be a direct consequence of the presence and operation of the DWPs.

Both the SPOT and TGL applicants conducted numerous modeling simulations of potential oil spill scenarios that could occur in upland, inshore, nearshore, and offshore locations. By far the worst-case scenario for effects to ESA-listed species under NMFS's jurisdiction (largest volume of oil spilled and aquatic area impacted) was labeled the Worst Credible Oil Spill Discharge. Because the areas affected by the Worst Credible Oil Spill Discharge scenario generally overlap and encompass the aquatic areas that might be affected by the other smaller spill scenarios, we determined that this was the appropriate scenario to analyze for an oil spill from the DWP projects.

The USCG conducts project specific independent risk assessments (IRA) for each deepwater port license application. Those assessment are contained in the EIS and associated appendices for each project (TGL DWP Biological Assessment, November 2020 and U.S. Coast Guard (2022). These project-specific IRAs contain specific risk profiles including oil spill probabilities for the worst credible scenarios for the proposed DWPs. While the probability of either DWP experiencing a Worst Credible Oil Spill Discharge is very low, such an oil spill could have significant consequences on ESA-listed species and critical habitat. Based on our past experience with oil pipelines in the Gulf of Mexico, we believe there is a sufficient degree of certitude associated with a Worst Credible Oil Spill Discharge that warrants analysis in order to be conservative for the benefit of ESA resources. We also believe that it is extremely unlikely that both DWPs would experience a Worst Credible Oil Spill Discharge during the 30-year life of the projects. We chose the largest volume discharge scenario (687,602 bbl of heavy crude [WCS] for the SPOT DWP) as the scenario to analyze for this Opinion because we do not expect effects to ESA-listed species and critical habitat to exceed the effects of that Worst Credible Oil Spill Discharge scenario. This determination is based on both the larger size of the potential discharge, as well as the fact that the TGL project does not propose to transport heavy crude (WCS). Due to the tendency of TGL's proposed oil types (WTI and WTI-Light crude) to evaporate more quickly, impacts from a spill of those types of crude oil would not have as far-reaching impacts as a spill involving heavy crude oil proposed for export from the SPOT DWP.

Worst Credible Oil Spill Discharge at the SPOT DWP

The "worst case discharge" for a marine transportation-related facility is the loss of the entire capacity of all in-line and break out tank(s) needed for the continuous operation of the pipelines used for the purposes of handling or transporting oil, in bulk, to or from a vessel regardless of the presence of secondary containment, plus the discharge from all piping carrying oil between the marine transfer manifold and the non-transportation-related portion of the facility, 33 CFR 154.1029(b). In accordance with this, the worst case discharge from the SPOT DWP calculated by taking into account the following:

- Rupture of both crude oil export pipelines leading to subsea oil spill, caused by a dropped or dragged anchor;
- Maximum time to isolation: the maximum time to discover the release from the pipeline plus the maximum time to shut down flow from the pipeline;
- Maximum flow rate: estimate based on the maximum system pressure, assumed to be 1.5 times the design maximum flow rate of 85,000 bph; and
- Discharge volume: maximum time to isolation × maximum flow rate + total line drainage volume.

Thus, the worst credible discharge scenario selected for oil fate and trajectory modeling is an underwater crude oil release from a simultaneous rupture of both 36-inch crude export pipelines at a nearshore location and close to the SPOT DWP (Figure 33, Figure 34, and Figure 35). The modeled release would occur in two phases: early phase and late phase. An early loss would occur over 30 minutes, resulting in a discharge volume of 63,750 bbl before a shutdown. This would be followed by late phase discharges (assumed to last up to 36 hours) after shutdown, involving a release volume of an additional 623,852 bbl from the line drainage. The total volume modeled was 687,602 bbl. For the worst case discharge ("WCD"), as defined by 33 CFR 154.1029, it is assumed that no response efforts take place to mitigate the impacts of the spill. While this assumption provides a worst case scenario for the volume and dimensions of a potential spill, NMFS is required to use "best available scientific and commercial information" to determine what is "reasonably certain to occur." Past experiences and mandatory response plans indicate that containment and clean-up efforts are extremely likely to occur in response to any large oil spill. While these response efforts may help to protect shorelines, impede the transport of oil, and remove oil from the affected environment, there are also some potential adverse effects to ESA-listed species that may result from response efforts, such as vessel strikes, direct injury from skimming and burning, and indirect impacts from dispersant use and other response activities. In order to ensure an analysis that is conservative towards ESA-listed species, we have assumed the oil spill volumes and dimensions defined as the WCD under 33 CFR 154.1029, but have also incorporated the potential adverse effects that may result from likely spill response efforts, by using the impact analysis developed in the DWH Programmatic Damage Assessment and Response Plan (PDARP) (which includes effects from response efforts) as the primary basis for our effects analysis. We also note that if the action agencies or another federal agency must approve the applicants' facility response plan or oil spill response plan, additional Section 7 consultation may be required at that time. Approval or authorization of any such plans would constitute a separate, future federal action that is not covered by this consultation.

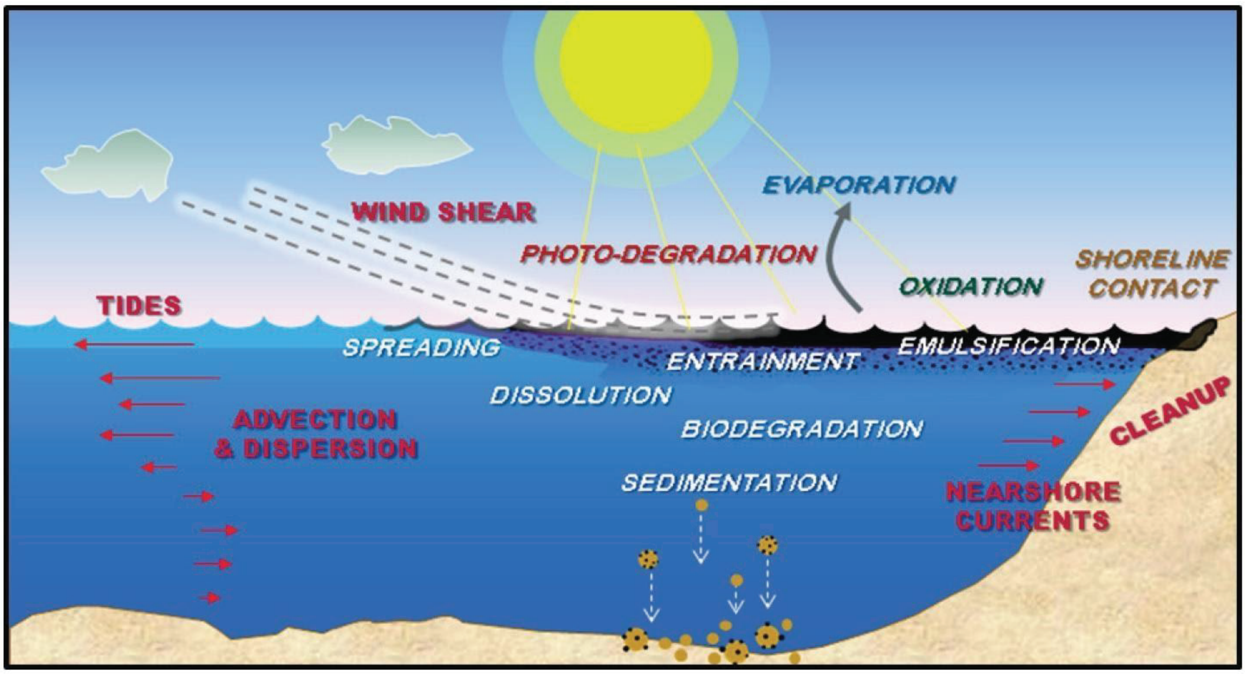

Figure 33. Oil Fate and Trajectory Modeling Overview (Figure 4.6-7 in the SPOT FEIS, July 2022)

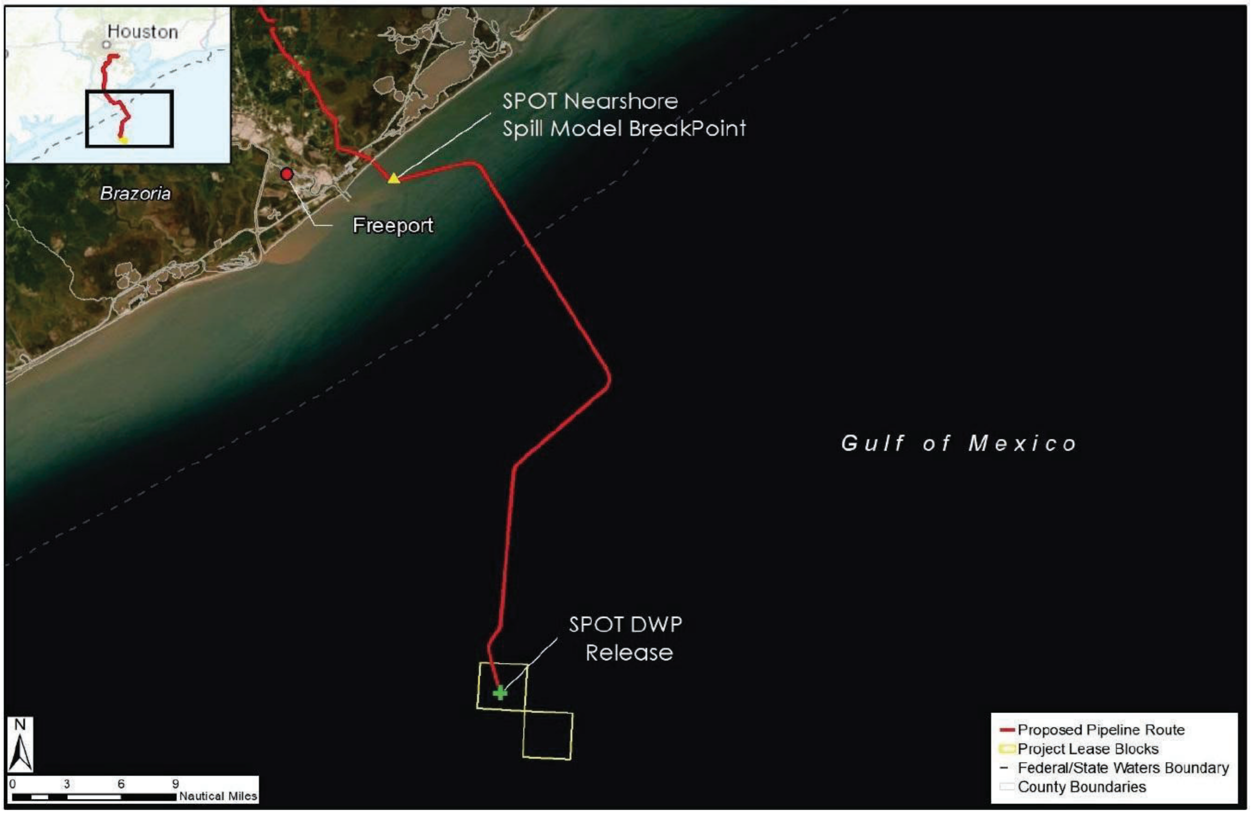

Figure 34. Selected Spill Locations: Nearshore and at the Deepwater Port (Figure 4.6-8 in the SPOT FEIS, July 2022)

Spill Fate and Trajectory Results of the SPOT DWP Worst Case Discharge
Potential impacts were examined regarding:

- The probability and locations of shorelines which could contact oil above a 100 grams per square meter (g/m^2) threshold for environment impacts and 1 g/m^2 for socioeconomic impacts; and
- The probability and locations of water surface areas where oil may float at or above 0.1 micrometer, the defined thickness threshold levels indicating oil may be visible.

Probabilistic results are presented as a composite of the range of possible locations to which oil may travel under the various conditions. Individual worst cases are then examined among the range of possible trajectories. In Scenario 1, the release was simulated to occur from a break at the SPOT DWP offshore platform (Figure 37 and Figure 40). In Scenario 2, the release occurred 2 miles off the coastline where the 36-in crude oil delivery pipelines first bend direction after running perpendicular to the coast (Figure 36 and Figure 39). In Scenario 3, the release was simulated to occur to the VLCC moored at the SPM buoy because of another vessel colliding with it (Figure 38 and Figure 41).

In the modeled oil spill events, two different simulations were run. The purpose of the first simulation was to understand how the potential oil spill impact propagation would affect the shoreline. The second simulation was used to predict how the oil would move with oceanic currents around the GoM and the total potential oiling impact on those areas.

Figure 39, Figure 40, and Figure 41 show the simulation results for each of the three scenarios for shoreline oiling for an ultralight crude oil (e.g., Condensate) spill. In 40 percent of the Scenario 2 events close to shore, the oil spill did not affect the shoreline, but turned and moved seaward for the next 14 days. However, in the other 60 percent of the cases, oil reached the shoreline around Freeport Texas.

The simulation of Scenario 1 shows that shoreline oiling (above a 5 percent probability) may occur along approximately 250 miles of shoreline between Port Aransas and Port Arthur, and and the simulation of Scenario 2 shows that shoreline oiling (above a 5 percent probability) may occur along 122 miles of shoreline between Port O'Connor and Galveston Island. The shorelines most likely to contact oil as a result of an ultralight crude oil spill in Scenarios 1 and 2 range from the region between Port O'Connor to Freeport, Texas. Figure 36 and Figure 37 show the GoM potential oil spill impacts of a spill in both locations. Scenario 2 (at the SPOT DWP) has a greater potential oil spill impact on more of the shoreline area and GoM because the offshore ocean currents carry the oil farther and currents close to shore are less strong and therefore do not have as great an effect as they do further offshore (Figure 38). In Scenario 1, a majority of the oil would go directly to the shoreline or stay in the same area when released close to shore. The three scenarios for the spill fate and trajectory modeling were simulated using the three different crude oil types that are proposed to be exported from the SPOT DWP (ultralight crude oil, e.g., Condensate; light crude oil, e.g., West Texas Intermediate [WTI]; and heavy grade crude oil, e.g., Western Canadian Select [WCS]).

The simulation of the shoreline oiling impact of Scenario 3, a VLCC collision resulting in an oil spill near the SPOT DWP SPM buoys (due to high traffic volumes in those areas) is depicted in Figure 38. The vessel collision spill scenario (scenario 3) simulates a release at the top meter of the water surface (3.3 feet) in the vicinity of the DWP. The three oils were simulated, and the entire release occurred in one phase. All three oils initially released at a volume of 614,285 bbl for 1.5 hours. The highest probability of surface oiling to the shoreline from Scenario 3 is near Freeport, but a small probability (5 percent) of surface oiling ranges from Port Aransas to Galveston Island. The ocean currents would take the oil spill towards the northwest, resulting in a 5 percent probability of surface oiling impact on Freeport and Galveston Island. Figure 41 shows the surface oil due to oceanic currents in the GoM.

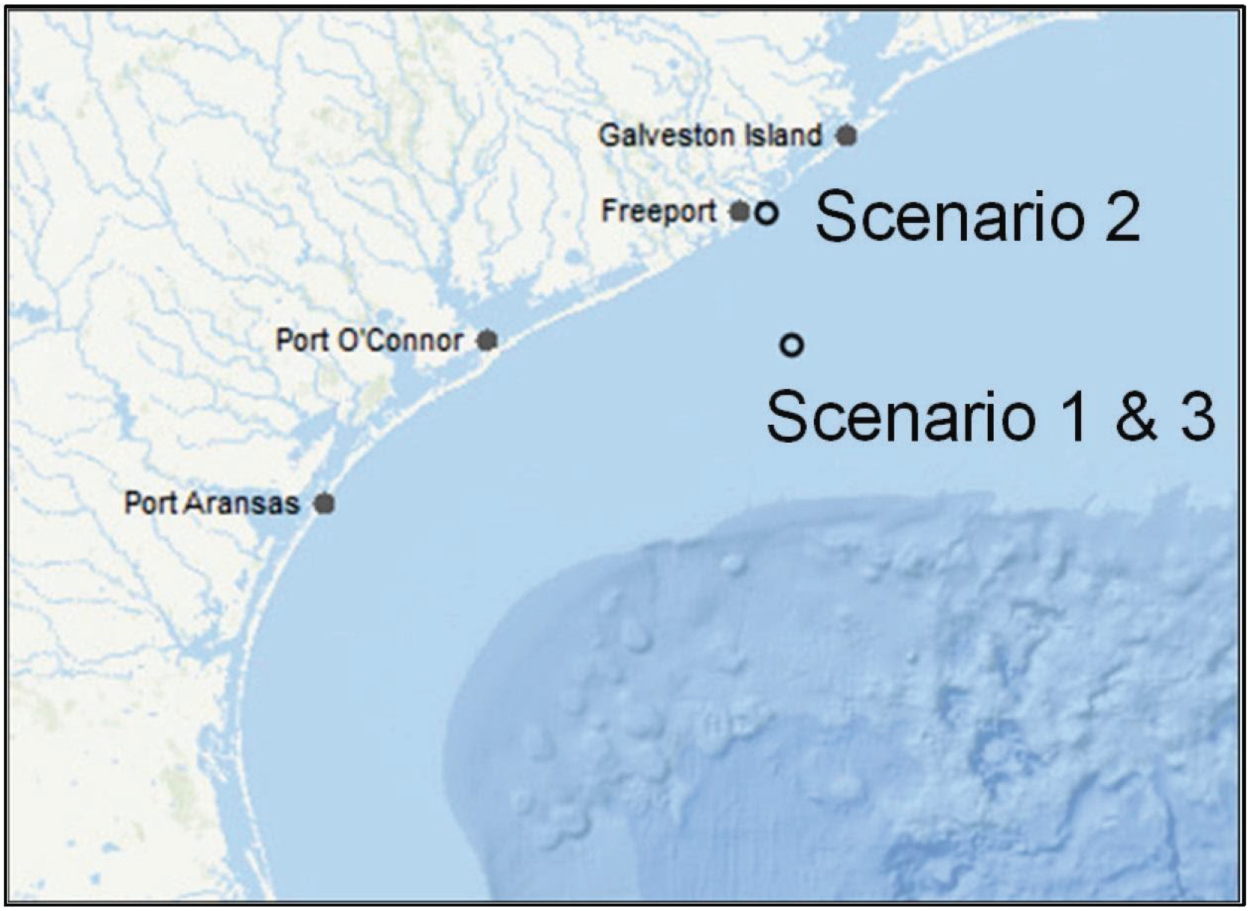

Figure 35. Locations of release points for Scenario 1 & 3 (SPOT DWP) and Scenario 2 (2 miles from shore) near Freeport Texas (Figure 4.6-9 in the SPOT FEIS, July 2022)

Figure 36. Potential Oil Spill Effect on Shoreline from a Double Pipeline Breach 2 Miles from Shore (Scenario 2, Figure 4.6-10 in the SPOT FEIS, July 2022)

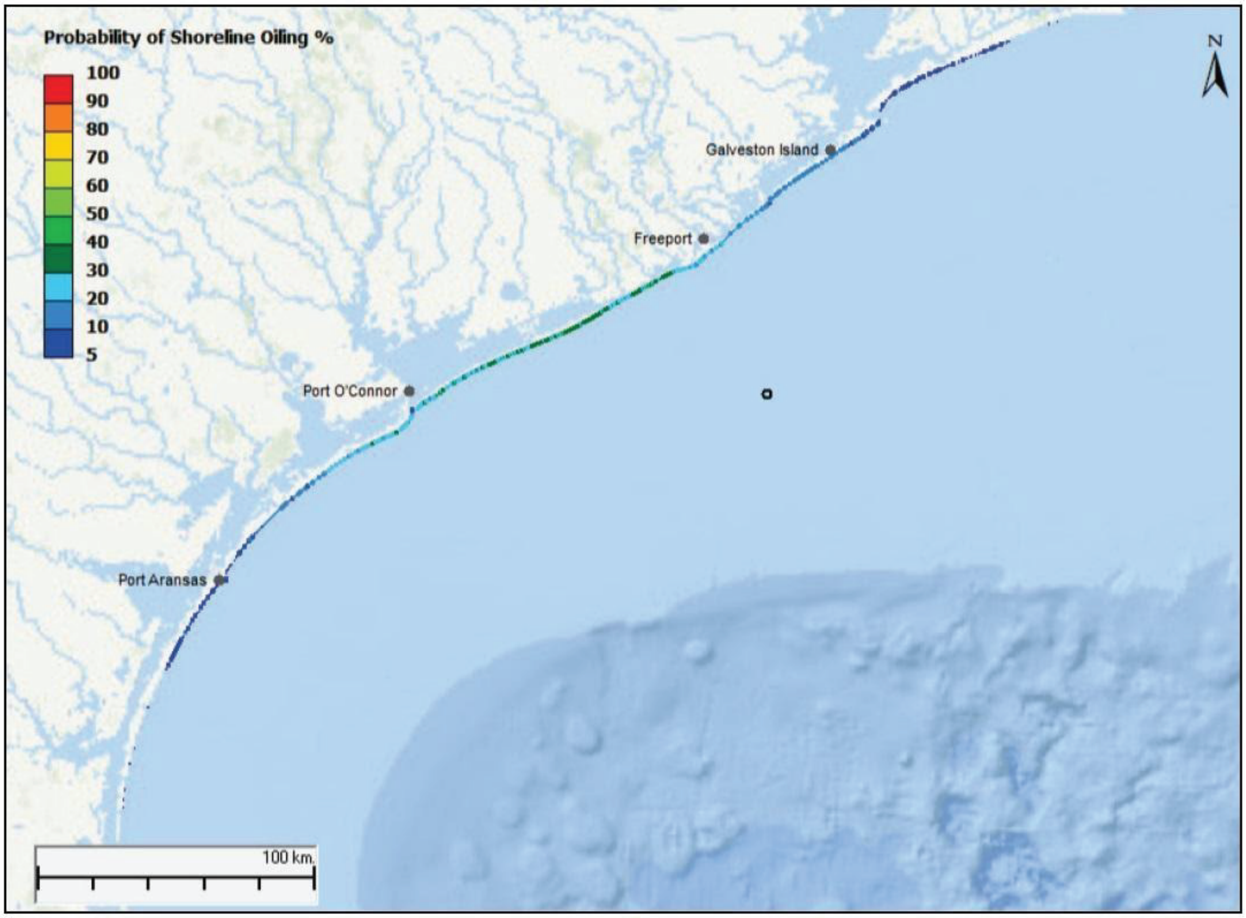

Figure 37. Potential Oil Spill Effect on Shoreline from a Double Pipeline Breach at the SPOT DWP (Scenario 1, Figure 4.6-11 in the SPOT FEIS, July 2022)

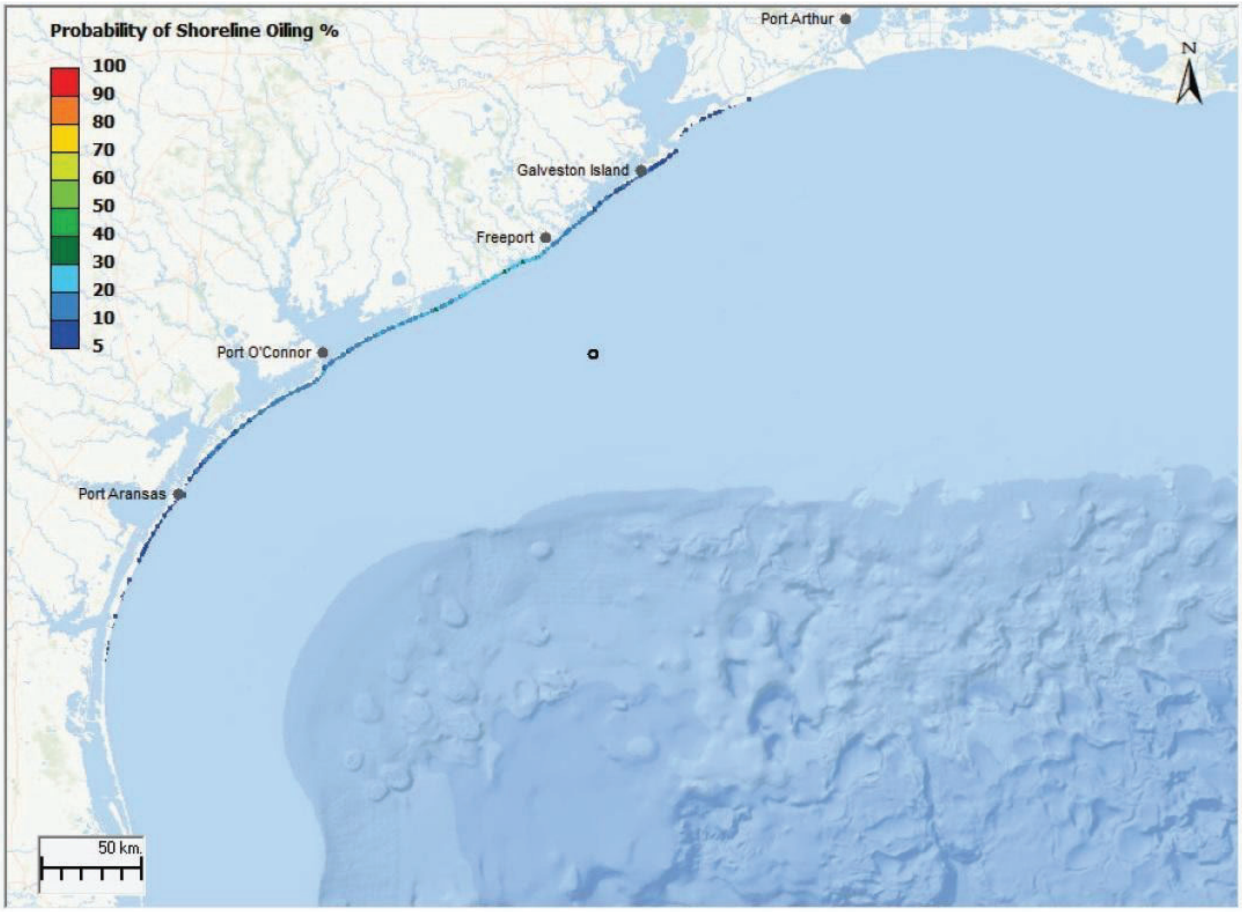

Figure 38. Potential Oil Spill Effect on Shoreline from a Full Release of Cargo from a VLCC at the SPOT DWP (Scenario 3, Figure 4.6-12 in the SPOT FEIS, July 2022)

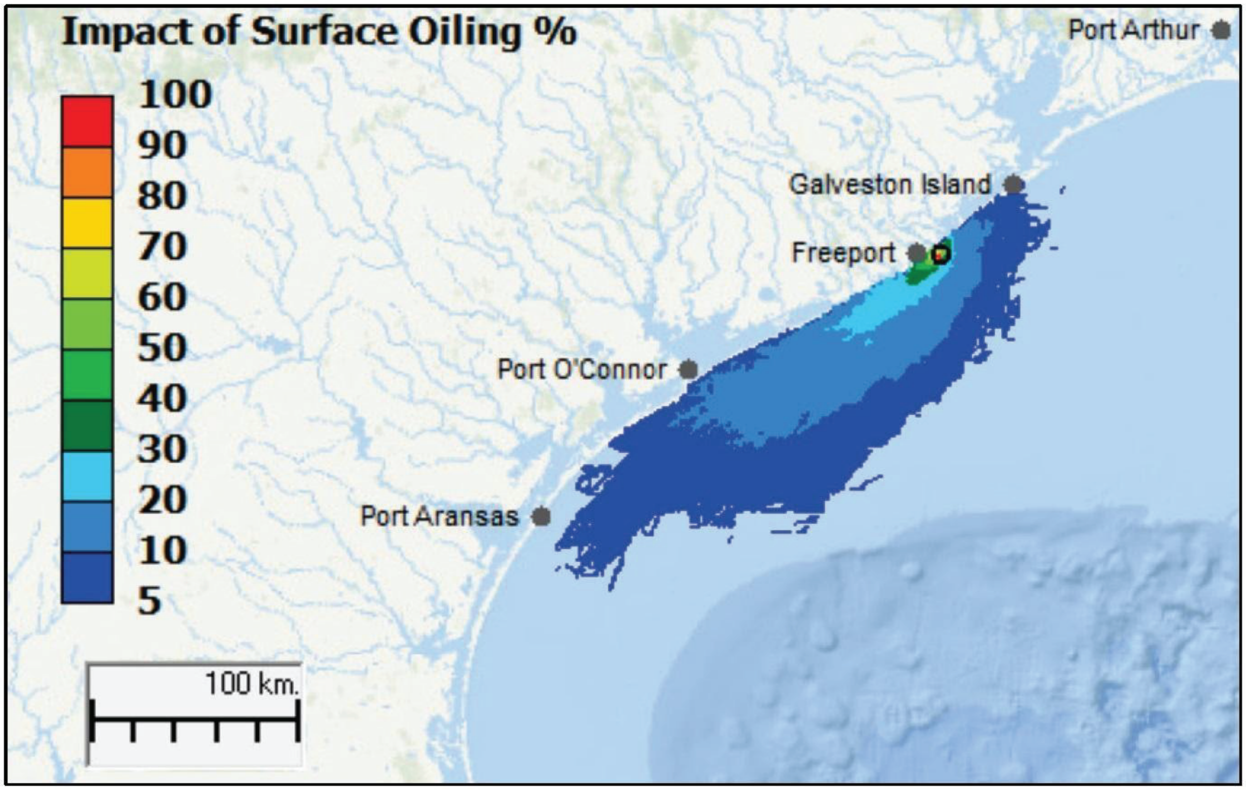

Figure 39. 2 Miles from Shoreline Scenario (Scenario 2) of Gulf of Mexico Potential Oil Spill Impact (Figure 4.6-13 in the SPOT FEIS, July 2022)

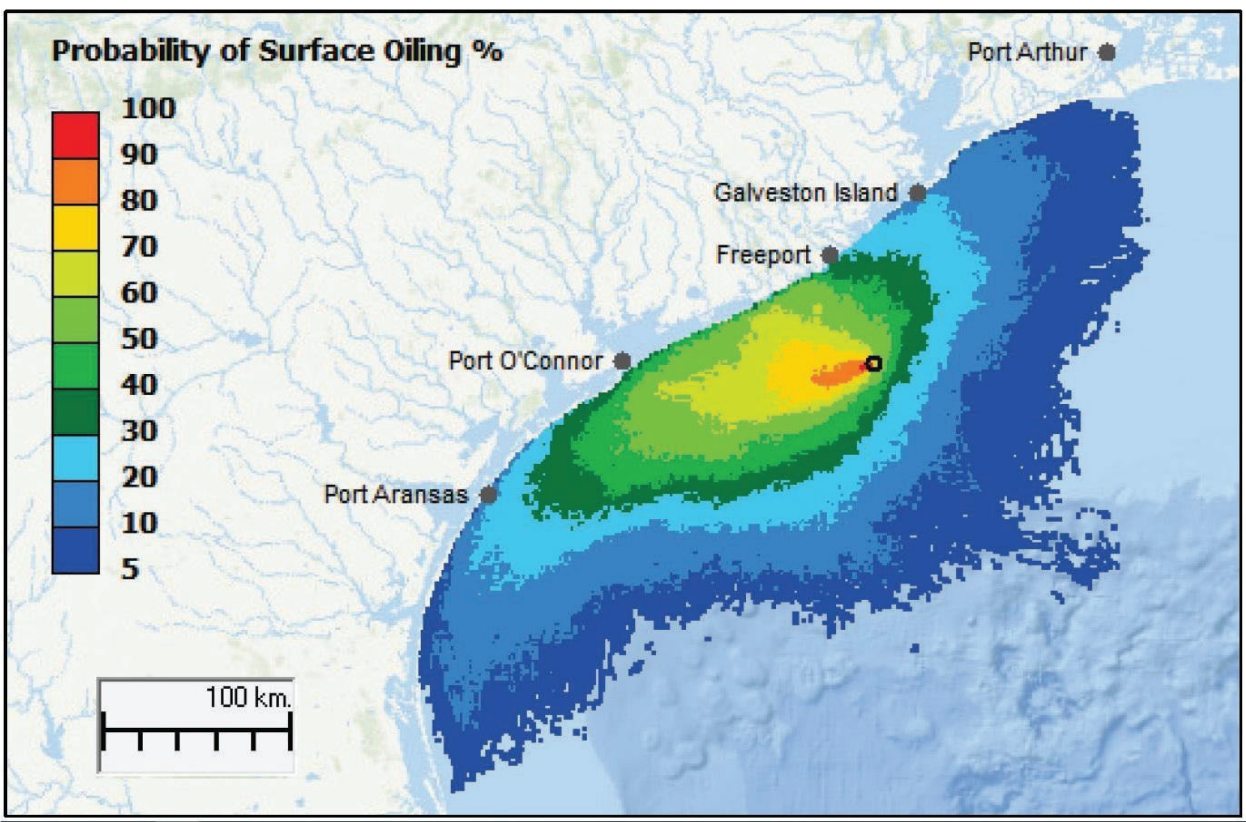

Figure 40. Pipeline Breach at SPOT DWP Scenario (Scenario 1) of Gulf of Mexico Potential Oil Spill Impact (Figure 4.6-14 in the SPOT FEIS, July 2022)

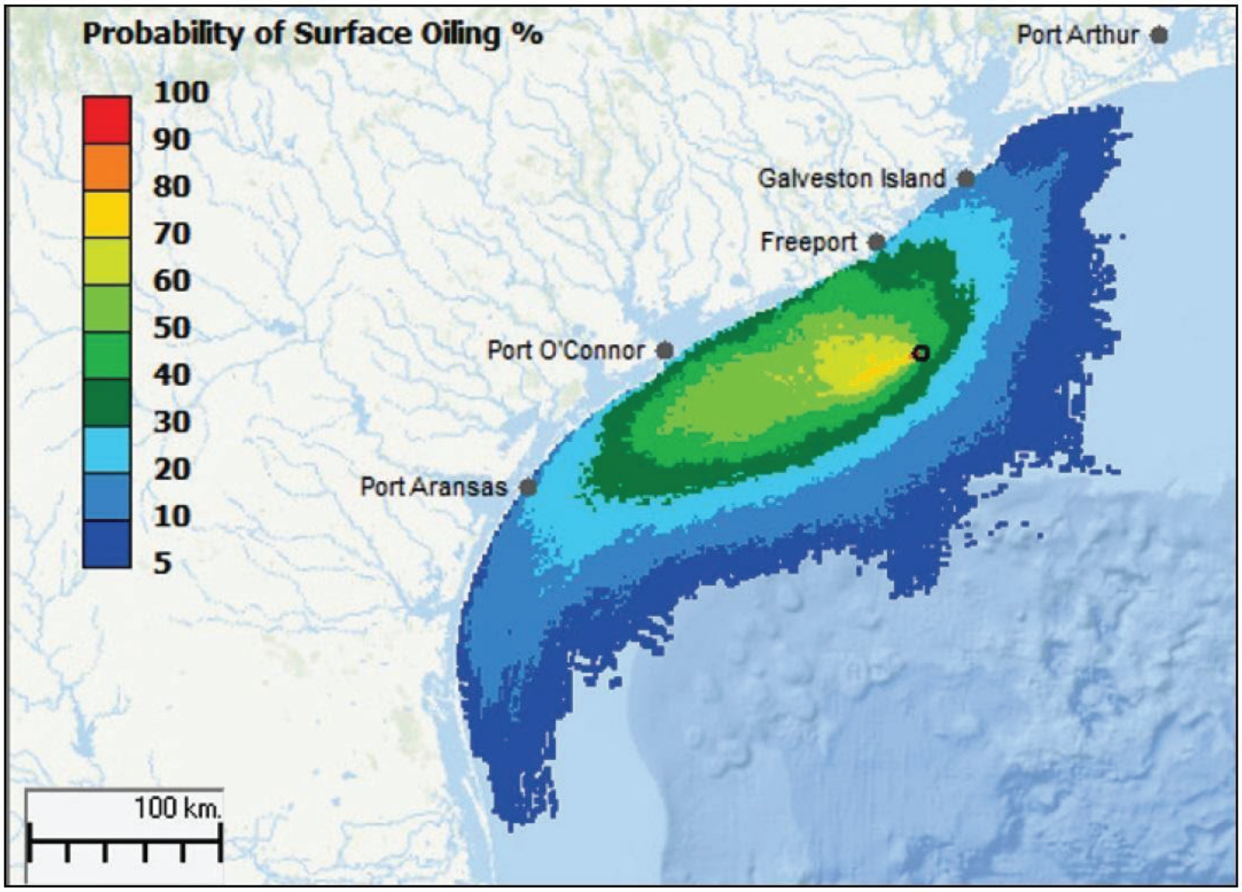

Figure 41. VLCC Scenario (Scenario 3) of Gulf of Mexico Potential Oil Spill Impact (Figure 4.6-15 in the SPOT FEIS, July 2022)

A Worst Credible Oil Spill Discharge from a SPOT pipeline failure could result in the release of up to 687,602 barrels (bbls) of oil from the two pipelines. The only oil spill greater than 10,000 bbl to occur in the last 20 years within the Gulf of Mexico was the DWH spill in 2010 (see details on the DWH spill effects in Section 4 above). The ecological effects of the DWH spill and disaster response were the subject of one of the largest scientific research efforts ever conducted in the Gulf, and the results of that extraordinary analytical effort were presented in the DWH Programmatic Damage Assessment and Response Plan (PDARP; DWH Trustees 2016). Given the fact that the DWH impacts occurred in the same general area (western Gulf of Mexico) as the action area for this Opinion, and the extreme complexity and difficulty in predicting the ecological impacts of a theoretical future spill, we have determined that the best information available to estimate the potential effects of an oil spill greater than 500,000 bbls resulting from the proposed actions is to use the impacts documented from the DWH spill (scaled down to the appropriate level). In our analysis of effects of such a spill, we assume that the spill response and cleanup efforts will be similar to those implemented during and after the DWH spill (scaled appropriately) and that the effects of those response activities will be comparable to those documented for the DWH spill. In terms of the volume of oil spilled, a

Worst Credible Oil Spill Discharge estimated for the proposed actions (687,602 bbls) is approximately 14% of the volume discharged during the DWH spill (4.9 million bbls).

The DWH spill and associated response activities (e.g., skimming, burning, and application of dispersants) resulted in adverse effects on ESA-listed sea turtles, sperm whales, and Rice's whales. Giant manta ray and oceanic whitetip sharks may also have been affected, but were not proposed to be listed until several years after the DWH oil spill and therefore impacts to these species were not analyzed in the DWH PDARP. The DWH oil spill extensively oiled vital foraging, migratory, and breeding habitats of ESA-listed species throughout much of the northern GoM. *Sargassum* habitats, benthic foraging habitats, surface and water column waters, and sea turtle nesting beaches were all affected by DWH. Sea turtles and whales were exposed to DWH oil in contaminated habitats; breathing oil droplets, oil vapors, and smoke; ingesting oil-contaminated water and prey; and by maternal transfer of oil compounds to developing embryos. Translocation of sea turtle eggs from the Gulf of Mexico to the Atlantic coast of Florida resulted in the loss of sea turtle hatchlings. Other response activities, including vessel strikes and dredging also resulted in documented turtle deaths.

5.1 Effects of Oil Spill on ESA-Listed Sea Turtles

NMFS (2020) estimates that as many 7,590 large juvenile and adult sea turtles (Kemp's ridleys, loggerheads, and unidentified hardshelled sea turtles), and up to 158,900 small juvenile sea turtles (Kemp's ridleys, green turtles, loggerheads, hawksbills, and hardshelled sea turtles not identified to species) were killed by the DWH oil spill (Table 15 and Table 16). Small juveniles were affected in the greatest numbers and suffered a higher mortality rate than large sea turtles. Leatherback foraging and migratory habitat was also affected and though impacts to leatherbacks were unquantified, it is likely some died as a result of the DWH spill and spill response (NMFS USFWS 2013; Trustees 2016).

Table 15. Oceanic Juvenile Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley	206,000	35,500	51,000	86,500
Loggerhead	29,800	2,070	8,310	10,400
Green	148,000	15,300	39,800	55,100
Hawksbill	8,560	595	2,390	2,990
Unidentified	9,960	1,310	2,600	3,910
Total	402,320	54,775	104,100	158,900

Table 16. Large Juveniles and Adult Sea Turtles Exposed and Killed (estimate) by the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley age 4+	21,000	1,700	950	2,700
Kemp's ridley age 3	990	380	30	410
Kemp's, all	22,000	2,100	980	3,100
Loggerhead	30,000	2,200	1,400	3,600
Unidentified	5,900	630	260	890
Total	57,900	4,930	2,640	7,590

The estimated effects to ESA-listed sea turtles resulting from the Worst Credible Oil Spill Discharge from the proposed actions scaled to 14% of the DWH spill are presented in Table 17 and Table 18 below.

Table 17. Estimates of Oceanic Juvenile Sea Turtles that May Be Exposed and Killed by a Potential Worst Credible Oil Spill Discharge from the Proposed Projects Scaled to 14% of the Effects of the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley	28,840	4,970	7,140	12,110
Loggerhead	4,172	290	1,164	1,456
Green	20,720	2,142	5,572	7,714
Hawksbill	1,199	84	335	419
Unidentified	1,395	184	364	548
Total	56,325	7,669	14,574	22,246

Table 18. Estimates of Large Juveniles and Adult Sea Turtles that May be Exposed and Killed by a Potential Worst Credible Oil Spill Discharge from the Proposed Projects Scaled to 14% of the Effects of the DWH Oil Spill (Trustees 2016; Wallace et al. 2015)

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley age 4+	2,940	238	133	378
Kemp's ridley age 3	139	54	5	58
Kemp's, all	3,080	294	138	434
Loggerhead	4,200	308	196	504
Unidentified	826	89	37	126
Total	8,106	691	370	1,063

5.2 Effects of Oil Spill on Sperm Whales

With regards to sperm whales, NOAA (2015) determined that 16 percent of the GoM sperm whale population (or about 262 whales) were exposed to DWH oil. Thirty-five percent of those whales (or approximately 92 whales) are estimated to have been killed. We estimate the effects from the Worst Credible Oil Spill Discharge from the proposed actions scaled to 14% the size of the DWH spill) would result in approximately 2.2 percent of the GoM sperm whale population (or about 37 whales) exposed to the oil. Thirty-five percent of those whales (or approximately 13 whales) would likely be killed.

5.3 Effects of Oil Spill on ESA-Listed Fish

Oceanic whitetip sharks and giant manta rays are likely to encounter oil, if the oil is present in the water column where individuals are feeding or swimming, or if an individual breaks the surface under a slick. While the densities of these two species are unknown, both species commonly occur in the GoM.

We expect that the effects of exposure to oil and oil response activities on oceanic whitetip sharks and giant manta rays to be similar because both species are elasmobranchs with similar physiology and are expected to be widely dispersed in the action area. These species are free-swimming, often in deeper, pelagic waters and may aspirate oil that is dispersed in the water column through their gill filaments. Giant manta rays could ingest oil by filter feeding and oceanic whitetip sharks could ingest oil by ingesting contaminated prey, such as fish and squid. Oil could contact the skin of these species beneath the water surface or when these animals breach the surface. Oil and dispersants could affect prey availability for oceanic whitetip sharks and giant manta rays. Those effects would be dependent on timing, size and location of the spill proximity to the prey.

Oceanic whitetip sharks are a highly migratory species that historically had high catch rates. The numbers of these sharks are thought to have greatly declined in the GoM and that occurrences are much rarer now than historically. The highest abundance of these sharks are believed to be in the deep central waters of the GoM, though more recently the pelagic longline fisheries have reported fewer encounters (Young et al. 2016a). As a result, the likelihood of any particular individual being in the area of a spill large enough to have oil remaining in the water column is very small, but some individuals found in the footprint of such a spill would likely be affected. Nonetheless, some oceanic whitetip sharks are likely to be exposed to oil, and those exposures are likely result in effects similar to other marine species, including fitness reduction, and possibly leading to mortality. Because there are no abundance estimates for oceanic whitetip sharks in the GoM, and there was no analysis of effects on this species in the DWH PDARP, we are not able to quantify an estimated number of oil spill exposures or mortalities for this species. Instead we will use the spatial area estimated to be affected by the worst case discharge scenarios as an ecological surrogate for the amount and extent of potential impacts to oceanic whitetip sharks.

Giant manta rays are regularly found at FGBNMS, as well as in both pelagic and shallower waters. There are no known breeding aggregations in the GoM. Thus, some giant manta rays are likely to be exposed to oil, and those exposures would likely result in effects similar to other marine species, including fitness reduction, and possibly leading to mortality. Because there are no abundance estimates for giant manta rays for the GoM beyond the approximately 70 individuals documented at FGBNMS, and there was no analysis of effects on this species in the DWH PDARP, we are not able to quantify an estimated number of oil spill exposures or mortalities for this species. Instead we will use the spatial area estimated to be affected by the Worst Credible Oil Spill Discharge scenarios as an ecological surrogate for the amount and extent of potential impacts to giant manta rays.

Modeling indicates that the total sea surface area that may be impacted by the Worst Credible Oil Spill Discharge from the SPOT DWP is roughly 47,000 km^2.

5.4 Effects of Oil Spills on Loggerhead Sargassum Critical Habitat LOGG-S-2

The four habitat features of *Sargassum* essential for the conservation of loggerhead sea turtles are: (1) convergence zones, surface-water downwelling areas, the margins of major boundary currents (Gulf Stream), and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for the optimal growth of *Sargassum* and inhabitance of loggerheads; (2) *Sargassum* in concentrations that support adequate prey abundance and cover; (3) available prey and other material associated with *Sargassum* habitat including, but not limited to, plants and cyanobacteria and animals native to the *Sargassum* community such as hydroids and copepods; and (4) sufficient water depth and proximity to available currents to ensure offshore transport (out of the surf zone), and foraging and cover requirements by *Sargassum* for post-hatchling loggerheads, i.e., >10 m depth. All four essential features or primary constitute elements (PCEs), must be present in an area for it to function as *Sargassum* critical habitat for loggerhead sea turtles. The loss of one essential feature will result in a total loss in the conservation function of the critical habitat in that area.

The proposed actions will not have any effect on the first and fourth essentials features, as oil spills would not affect suitable water temperatures, water depths, convergence zones, downwelling, and other water current features. A Worst Credible Oil Spill Discharge and oil spill response activities from the proposed actions are likely to adversely affect the second and third essential features of the loggerhead sea turtle *Sargassum* critical habitat unit LOGG-S-2.

Physical processes, such as convergent currents and fronts that play a role in transporting, retaining, and concentrating *Sargassum*, are the same processes that act to concentrate oil. These physical processes would tend to increase the exposure of *Sargassum* and associated organisms to an oil spill (Trustees 2016). Powers et al. (2013) identified three pathways in which an oil spill can adversely affect *Sargassum* dependent communities:

- *Sargassum* patches can accumulate oil on the surface exposing animals to high concentrations of contaminants;
- application of dispersant in response to an oil spill can sink *Sargassum*, thus removing the habitat and potentially transporting oil and dispersant vertically; and
- low oxygen conditions can develop within the *Sargassum* habitat potentially stressing animals that reside in the alga.

Thus, we believe these pathways are likely to adversely affect the third PCE of loggerhead sea turtle *Sargassum* critical habitat unit LOGG-S-2, namely, available prey and other material associated with *Sargassum* habitat including, but not limited to, plants and cyanobacteria and animals native to the *Sargassum* community such as hydroids and copepods.

Sinking, burning or otherwise removing large numbers of individual clumps, patches, or lines of *Sargassum* could also reduce habitat density to the point where it can no longer support adequate prey abundance and cover. Thus, we believe activities taken in response to an oil spill are likely to adversely affect the second PCE of loggerhead sea turtle *Sargassum* critical habitat unit LOGG-S-2, namely, *Sargassum* in concentrations that support adequate prey abundance and cover.

The DWH oil spill resulted in an estimated 843 km^2 to 1749 km^2 of *Sargassum* areas with surface oiling (Hu et al. 2016). The Trustees (2016) estimated approximately 23 percent of the *Sargassum* in the northern Gulf of Mexico (at the time of the spill) was lost due to direct exposure to DWH oil on the ocean surface. Floating *Sargassum* samples collected up to 100 miles from the wellhead were shown to have been impacted by DHW oil (McDonald and Powers 2015). An additional measure of adverse effects to *Sargassum* from oil spills is the foregone surface area due to lost growth caused by oil exposure. An estimated 6,958 km^2 of *Sargassum* surface area was foregone as a result of the DWH oil spill. The loss of *Sargassum* habitat resulting from DWH was likely exacerbated by the use of oil dispersants (Powers et al. 2013).

Unlike loggerhead nearshore reproductive critical habitat, which is likely to be exposed to oil from larger offshore spills only, *Sargassum* floating in offshore areas could be exposed to oil from smaller spills as well as larger spills. Small-scale oil spills, which occur more frequently in the northern Gulf of Mexico, could affect localized *Sargassum* communities. However, we expect that patches exposed to such small-scale oil spills will recover quickly. An extremely large spill, such as DWH, would likely result in widespread, sea-scape level impacts that could make it difficult for juvenile turtles to locate suitable *Sargassum* habitat, particularly if dispersants are used in the aftermath of such a spill.

Containment of *Sargassum* patches within booms or skimmers would result in some reduction of patch concentration and prey availability. Dispersants could cause sinking of patches and directly affect prey abundance. In-situ burning could also cause destruction of patches.

We believe that oil spills and actions taken to respond to an extremely large oil spill at either of the proposed DWPs are likely to adversely affect *Sargassum* in concentrations that support adequate prey abundance and cover (PCE No. 2) and available prey and other material associated with *Sargassum* habitat including, but not limited to, plants and cyanobacteria and animals native to the *Sargassum* community such as hydroids and copepods (PCE No. 3). Immediate effects of *Sargassum* exposure to oil and oil and dispersant mixtures will likely include reduced prey abundance, reduced cover, and reduced developmental and foraging habitat.

To fully assess both the short-term and long-term effects of oil spills on the essential features of *Sargassum* critical habitat, we need to consider aspects of the algae's life cycle including seasonal movements and drift rate within the action area, growth rate, longevity, and resiliency to environmental disturbances. Unlike more fixed types of critical habitat (e.g., river stretches, or nesting beaches), *Sargassum* is a highly mobile habitat. The patterns of movement of *Sargassum* over many months is relatively consistent from year to year, and can be explained by prevailing surface currents and winds. Satellite data from 2003 to 2007 indicate that *Sargassum* starts growing each year in the Gulf of Mexico around March, and dies about a year later in the Atlantic in the area northeast of the Bahamas (Gower and King 2011). The rapid increase in the amount of *Sargassum* in the northwest Gulf each year from March to July strongly suggests that the Gulf of Mexico is the dominant source of new *Sargassum* growth, which occurs cyclically. Satellite data indicates strong growth early in the year in the Gulf of Mexico, with *Sargassum* advected by the Loop Current and Gulf Stream into the Atlantic each year in July and August (Gower and King 2011). *Sargassum* species in the northern Gulf of Mexico grow at an estimated rate of four percent per day (Lapointe 1986).

The amount of *Sargassum* exposed to an oil spill within the action area will depend, to a large extent, on the time of year given the seasonality and cyclical movement of *Sargassum* in the northern Gulf of Mexico. Continuous exposure of a particular *Sargassum* patch to oil could last days, weeks, or months depending on the size and location of the spill and other factors (e.g., wind speed and direction, season, and type of oil). For example, the full range of area affected by the DWH oil spill covered 26,025 to 45,825 km^2 (Trustees 2016). More heavily oiled patches that are closer to the spill source at the time of the spill, and areas exposed to both oil and oil dispersants, will likely die-off or sink to the ocean bottom. For example, DWH oil impacted floating *Sargassum* samples collected up to 100 miles from the wellsite (Trustees 2016).

Given its fast growth rate, continuous motion, and somewhat ephemeral nature, we expect a relatively high turnover rate for *Sargassum* patches under normal conditions. *Sargassum* habitat that is lost due to an oil spill will likely be replaced over time by the combination of movement by unexposed (or lightly exposed) existing patches and through new growth. While the adverse effects of a major oil spill on *Sargassum* communities within a given annual life cycle (described above) are well documented, the longer-term impacts in subsequent years or decades are not known. Although nearly one-quarter of all *Sargassum* habitat in the northern Gulf of Mexico was heavily exposed to oil after the 2010 DWH spill, follow-up aerial surveys in 2011 and 2012 documented a four-fold increase in *Sargassum* abundance following DWH. These results suggest that *Sargassum* can repopulate in the Gulf of Mexico within a year or two of an extremely large oil spill.

Based on a Geographic Information System comparison of the modeled Worst Credible Oil Spill Discharge scenarios to the LOGG-S-2 location, we estimate a Worst Credible Oil Spill Discharge resulting from the proposed actions could impact roughly 47,000 km^2 of LOGG-S-2 critical habitat. This would equate to approximately 12% of the 393,053.75 km^2 area designated as the LOGG-S-2 critical habitat unit.

In summary, we believe that an oil spill and actions taken to respond to a major oil spill resulting from the proposed actions are likely to adversely affect essential physical and biological features of loggerhead *Sargassum* critical habitat (i.e., concentrations of *Sargassum* habitat and available prey and other material associated with *Sargassum* habitat). An extremely large oil spill from either DWP would likely have detrimental effects to *Sargassum* communities that juvenile turtles depend on for food and shelter. The effects of oil exposure on *Sargassum* critical habitat would be severe and last for days, weeks or even months in the case of a major oil spill. However, the ephemeral nature and annual cycle of rapid growth, movement, and subsequent die-off, allows *Sargassum* to repopulate in the Gulf of Mexico in the years subsequent to an extremely large oil spill. Therefore, overall we do not expect an oil spill to affect this critical habitat unit's long-term ability to support adequate prey abundance and cover for loggerhead turtles.

6 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, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this Opinion (50 CFR 402.02).

During this consultation, we searched for information on future state, tribal, local, or private (non-federal) actions reasonably certain to occur in the action area that would have effects to sperm whales, green sea turtles (NA and SA DPSs), Kemp's ridley sea turtles, leatherback sea turtles, loggerhead sea turtles (NWA DPS), hawksbill sea turtles, oceanic whitetip sharks, giant manta ray and loggerhead sea turtle designated critical habitat Unit LOGG-S-2. We did not find any information about non-federal actions being planned or under development in the action area other than the actions described in the *Environmental Baseline* (Section 4), which we expect will continue in the future. Non-federal activities anticipated to continue into the future include commercial and recreational fishing, oil and gas activities, scientific research, marine noise generating activities, activities affecting climate change, and activities generating marine debris and pollution, as well as the vessel traffic associated with these activities.

An increase in these activities could increase the effects of these activities on ESA-listed resources. For some of these activities, an increase in the future is considered reasonably certain to occur. Given current trends in global population growth, threats associated with climate change, pollution, fisheries, vessel strikes and approaches, and sound are likely to continue to increase in the future, although any increase in effect may be somewhat countered by an increase in conservation and management activities. In contrast, more historic threats such as whaling and sea turtle harvest are likely to remain low or potentially decrease. For the remaining activities and associated threats identified in the *Environmental Baseline*, and other unforeseen threats, the magnitude of increase and the significance of any anticipated effects remain unknown. The best scientific and commercial data available provide little specific information on any long-term effects of these potential sources of disturbance on ESA-listed species. Thus, this consultation assumes effects in the future from ongoing human activities within the action area will be similar to those in the past and, therefore, are reflected in the anticipated trends described in the *Status of Species and Critical Habitat* and *Environmental Baseline* (Section 3 and Section 4).

7 INTEGRATION AND SYNTHESIS

7.1 Jeopardy Analysis

The analyses conducted in the previous sections of this Opinion serve to provide a basis to determine whether the proposed actions are likely to jeopardize the continued existence of sperm whale, green sea turtles (NA and SA DPSs), Kemp's ridley sea turtles, leatherback sea turtles, loggerhead sea turtles (NWA DPS), and hawksbill turtles, giant manta rays, and oceanic whitetip shark. In the Effects of the Action (Section 5.0), we outlined how the proposed actions would likely affect these species at the individual level and the extent of those effects in terms of the number of associated interactions, captures, and mortalities of each species to the extent possible based on the best available data. Now we assess each of these species' responses to these impacts, in terms of overall population effects, and whether those effects of the proposed actions, when considered in the context of the Status of the Species (Section 3.0), the Environmental Baseline (Section 4.0), and the Cumulative Effects (Section 6.0), are likely to jeopardize the continued existence of the ESA-listed species in the wild.

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

The NMFS and USFWS's ESA Section 7 Handbook (USFWS and NMFS 1998) defines survival and recovery, as they apply to the ESA's jeopardy standard. Survival means "the species' persistence…beyond the conditions leading to its endangerment, with sufficient resilience to allow recovery from endangerment." 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. Recovery means "improvement in the status of a listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the Act." Recovery is the process by which species' ecosystems are restored or threats to the species are removed so self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

For any species listed globally, our jeopardy determination must find the proposed actions will appreciably reduce the likelihood of survival and recovery at the global species range. For any species listed as DPSs, a jeopardy determination must find the proposed actions will appreciably reduce the likelihood of survival and recovery of that DPS.

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and their designated critical habitat as a result of implementing the proposed actions. This synthesis incorporates conservation measures described in the *Description of the Proposed Actions* (section 2.1) and in section 9.2, *Effect(s) of the Take*.

7.1.1 Sperm Whales

Sperm whales are the only ESA-listed marine mammal species under NMFS's jurisdiction that is likely to be found within the action area. In this section, we detail our integration and synthesis of effects for this species, in which we rely on, and summarize information, presented in the *Species Status*, the *Environmental Baseline*, the *Cumulative Effects,* and the *Effects of the Action* Sections of this Opinion.

Sperm whales occur in all oceans of the world. The best estimate of the current worldwide abundance of sperm whales is between 300,000 and 450,000 individuals (Whitehead 2002). Within the Atlantic, their abundance is estimated at 90,000 to 134,000 individuals and within the GoM, there are between 763 (NMFS 2015) and 2,128 (Roberts et al. 2016) resident whales. While there are no long-term estimates of abundance trends within the GoM, sperm whales in this region are thought to have been heavily impacted by the DWH oil spill, which may have resulted in a population decline (Chiquet et al. 2013). Sperm whales are still likely one of the most abundant large whale species, and on a global scale they show little genetic differentiation in terms of nDNA likely due to male sperm whales roaming widely. Within ocean basins, and even more so within semi-enclosed basins such as the GoM, sperm whales do show some genetic differentiation based on mtDNA. This differentiation is thought to be the consequence of shorter-ranging, and in some cases resident, females.

Sperm whales in the GoM were impacted by the 2010 DWH oil spill, with an estimated 262 individuals exposed to DWH oil and approximately 92 whales dying from that exposure or other effects of the spill and response efforts (NOAA 2015). Long-term impacts to fitness and reproduction from ongoing exposure to remnant oil from the spill may continue to this day. Future effects on sperm whales from oil spills from BOEM regulated oil and gas activities in the GoM were estimated in NOAA (2020) to result in 712 exposures (over 50 years) which will likely result in a range of responses including mortality (at least one), injury, reproductive failure, impairment, and harassment. Other ongoing routes of effects to this species, including noise, vessel strikes, fisheries impacts and habitat degradation, are expected to continue in the future at similar levels.

We estimate that a Worst Credible Oil Spill Discharge from the proposed actions could expose up to 37 sperm whales to spilled oil, and approximately 13 of those whales would likely be killed by the oil or the oil spill response activities (dispersants, in situ burning, vessel strikes, etc.). In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of sperm whales, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

In the GoM, there are estimated to be between 763 and 2,128 individual sperm whales (NOAA 2020). At a maximum, 2,128 represents 2% of all sperm whales in the Atlantic Ocean basin and less than 1% of the species abundance globally. The estimated adverse effects from the proposed actions to the sperm whales located in the western GoM (37 exposures and 13 deaths) are modest compared to the overall size of the listed species' population (300,000 – 450,000 worldwide; (Whitehead 2002). Sperm whales in the GoM are primarily composed of resident maternally-related groups of females and juveniles. Male sperm whales tend to be wide ranging and not resident to any particular area. Given the global connectedness of sperm whale populations, genetic similarities across subpopulations, and the small percentage the affected GoM subpopulation represents globally, we find that the proposed action is not likely to appreciably reduce the likelihood of the survival of the species in the wild.

Recovery

The recovery plan for sperm whales (NMFS et al. 2011b) lists the following recovery objectives:

Objective 1: Achieve sufficient and viable populations in all ocean basins

Criterion: Given current and projected threats and environmental conditions, the sperm whale population in each ocean basin in which it occurs (Atlantic Ocean/Mediterranean Sea, Pacific Ocean, and Indian Ocean) satisfies the risk analysis standard for unlisted status (has less than a 10% probability of becoming endangered, and has no more than a 1% chance of extinction in the next 100 years). Any factors or circumstances that are thought to substantially contribute to a real risk of extinction that cannot be incorporated into a Population Viability Analysis will be carefully considered before delisting takes place.

Objective 2: Ensure significant threats are addressed

Criteria: Factors that may limit population growth (those that are identified in the threats analysis as high or medium or unknown) have been identified and are being or have been addressed to the extent that they allow for continued growth of populations.

Regarding Objective 1, the relatively small number sperm whales that may be impacted or killed as a result of the proposed actions is not expected to have any measurable effect on the size or viability of the Atlantic Ocean population of sperm whales, which is estimated at 90,000-134,000 individuals based on the most recent available information. Up to 37 exposures and 13 mortalities, along with the associated reductions in future reproduction from a potential oil spill would effect a small proportion of the total population for a single breeding season (assuming spill response efforts are wrapped up within 1 year). This level and duration of effects would not be expected to produce a measurable impact on the recovery potential of a population of this size and broad distribution.

Regarding Objective 2, there are no limiting factors identified as high or medium level threats in the Recovery Plan, but there are several identified as "unknown." Anthropogenic Noise, Loss of Prey Base Due to Climate Change, and Contaminants and Pollutants are listed as unknowns throughout the species' range. Of these three factors, the only one that is directly relevant to this analysis is Contaminants and Pollutants. One of the criteria listed as necessary to bring the threat level for this factor down to "low" is:

Effects of oil spills and contaminants are determined to not affect the potential for continued growth or maintenance of the sperm whale population and actions taken or having been taken to minimize potential effects have been proven effective.

The massive interagency coordination effort and subsequent development of preemptive spill response plans associated with the DWH spill produced significant advances in addressing the threats posed by future oil spills to sperm whales in the Gulf of Mexico (and elsewhere). We believe the effects of a potential oil spill resulting from the proposed actions would not interfered with the recovery objectives above, and are not likely to result in an appreciable reduction in the likelihood of sperm whales' recovery in the wild.

Based on the full analysis provided above, we believe that the effects of the proposed actions are not likely to cause an appreciable reduction in the likelihood of survival and recovery of sperm whales, and are not likely to result in jeopardy to the species.

7.1.2 Sea Turtles

As discussed in the *Status of Species* and *Environmental Baseline* sections, the major anthropogenic stressors that contributed to the sharp decline of ESA-listed sea turtle populations in the past include habitat degradation, direct harvest, commercial fisheries bycatch, and marine debris. While sea turtle populations are still at risk, efforts made over the past few decades to reduce the impact of these threats have slowed the rate of decline for many populations. Bycatch reduction devices have reduced the incidental take of sea turtles in many commercial U.S. fisheries. Turtle excluder devices, which are required in federal shrimp trawl fisheries, are estimated to have reduced mortality of sea turtles by approximately 95% (NMFS 2014). Mitigation measures required in other federal and state fisheries (e.g., gill net, pelagic longline, pound nets) have also resulted in reduced sea turtle interactions and mortality rates. Increased conservation awareness at the international scale has led to greater global protection of sea turtles. While vessel strikes, power plants, dredging, pollutants, and oil spills still represent sources of mortality, sea turtle mortalities resulting from these activities within the action area are expected to either remain at current levels, or decrease with additional research efforts, conservation measures, and the continued implementation of existing environmental regulations. Based on our *Cumulative Effects* analysis (Section 6), some current threats to sea turtles are expected to increase in the future. These threats include global climate change, oil and gas development, marine debris, and habitat degradation. However, predicting the magnitude of these types of threats in the future or their impact on sea turtle populations is difficult.

All sea turtle life stages are important to the survival and recovery of the species but one life stage may not be equivalent to other life stages. For example, the take of male juveniles may affect survivorship and recruitment rates into the reproductive population in any given year, but is unlikely to significantly reduce the reproductive potential of the population. For sea turtles, a very low percent of hatchlings is typically expected to survive to reproductive age. Therefore, the loss of hatchlings from a population level standpoint is not as significant with respect to the survival and recovery of the species as the loss of sexually mature life stages. The death of mature, breeding females can have an immediate effect on the reproductive rate of the species. Sublethal effects on adult females may also reduce reproduction by hindering foraging success, as sufficient energy reserves are necessary for producing multiple clutches of eggs in a breeding year.

In our *Effects Analysis* (Section 5), we determined that exposure to oil spills and spill response activities (dispersants, in situ burning, vessel strikes, etc.) resulting from the proposed actions are likely to adversely affect ESA-listed sea turtles. Oil spills associated with the proposed actions would likely have both lethal and sublethal effects on a large number of sea turtles within the action area. The anticipated effects on sea turtles exposed to oil and spill response activities (e.g., dispersants, in situ burning, vessel strikes, etc.) range from minor to severe, depending on the spill volume and exposure level of individual turtles. Effects on sea turtles from light, short-term exposure to oil include moderate irritation to eyes, skin, and respiratory organs, and incidental oil ingestion. Moderate to high levels of oil exposure are more likely to result in significant fitness consequences for individual sea turtles exposed. Anticipated effects include impairment of feeding, swimming, and mating behaviors, high degree of irritation to eyes, ears, and respiratory structures, and ingestion of large amounts of oil. Impacts from spill response activities can also range in severity from minor irritation due to exposure to dispersant chemicals, to mortality from response-vessel strikes and in-situ burning of oil. Sea turtles can also be adversely affected by the loss of *Sargassum* and benthic habitats used for forarging and shelter, that are contaminated by spilled oil.

Different age classes may experience varying rates of mortality and resilience. We summarize the combined effects of the proposed actions by type of effect (i.e., sublethal exposure and lethal exposure) and life stage (i.e., adults/neritic juveniles, and smaller oceanic juvenile) in the tables above for each sea turtle species or DPS (Table 17 and Table 18). Mortality includes severely oiled turtles (that are expected to eventually die), turtles struck and killed by response-vessels, and all other stressors resulting in direct sea turtle mortality. Sublethal physical injury includes moderate to high oil exposure and other impacts that could lead to temporary fitness consequences, but are not expected to result in mortality.

The effects of the DWH spill, as documented in the DWH PDARP (DWH Trustees 2016), are used to project the effects of a Worst Credible Oil Spill Discharge from the DWPs. In terms of the volume of the oil spilled, a Worst Credible Oil Spill Discharge from SPOT (687,602 bbls) would be approximately 14% of the volume of the DWH spill (4.9 million bbls).

7.1.2.1 Green Sea Turtle

The green sea turtle was initially 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 May 6, 2016, the species was reclassified into 11 DPSs, including the NA DPS and the SA DPS, which were both listed as threatened. As noted in the *Status of Species* Section 3.2, information suggests that the vast majority of the anticipated green sea turtles in the Gulf of Mexico and South Atlantic regions are likely to come from the North Atlantic DPS. However, it is possible that animals from the South Atlantic DPS could be affected by the proposed actions. Based on Foley et al. (2007), we expect that approximately 96% of the green sea turtles in the GoM are part of the NA DPS, and the other 4% are part of the SA DPS. For these reasons, we conduct 2 jeopardy analyses (1 for each DPS). The NA DPS analysis assumes that 96% of animals adversely affected during the proposed actions are from that DPS. The SA DPS analysis assumes that 4% of the green sea turtles adversely affected by the proposed action are from that DPS.

Green Sea Turtle NA DPS

For this assessment, we assume the effects of a worst case discharge from the proposed actions would result in 14% of the exposures and mortalities of green sea turtles estimated for the DWH spill. The DWH PDARP assessment did not differentiate between the NA and SA DPSs of green sea turtles, but as explained above, we assume that 96% of the green sea turtle impacts estimated in the DWH PDARP were NA DPS green sea turtles. The DWH PDARP estimated the number of oceanic juvenile green sea turtles exposed to, and killed by, the DWH spill and oil spill response activities, but did not identify any neritic (near-shore, age 3+) juveniles or adults that might have been effected. We conservatively assume that all of the unidentified neritic juveniles and adults (Table 17 and Table 18) were green sea turtles in this analysis. We summarize the effects of a worstcase discharge and oil spill response activities resulting from the proposed actions on the green sea turtle NA DPS by type of effect (exposure and mortality) and life stage (neritic age 3+ and oceanic juvenile) in Table 19 below.

Table 19. Summary of the Effects of a Worst Credible Oil Spill Discharge on NA DPS of Green Sea Turtles

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
NA Green Oceanic Juvenile	19,891	2,056	5,349	7,405
NA Green Age 3+	793	85	36	121
Total	20,684	2,141	5,385	7,526

In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of NA DPS green sea turtles, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival
We estimate that the proposed actions could expose up to 20,684 green sea turtles from the NA DPS to oil and oil spill response activities over the 30-year life of the projects. Of those 20,684 turtles, we estimate that 7,526 of them willsuffer mortality, and that the remaining 13,158 will experience temporary effects and fully recover from these effects. These non-lethal exposures are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individuals suffering non-lethal injuries or stresses are expected to fully recover such that effects to these individuals would not result in reductions in reproduction or numbers of NA DPS green sea turtles. Any exposed turtles would be expected to stay within the same general area and no change in the distribution of NA DPS green sea turtles would be anticipated as a result of the projected non-lethal exposures.

The potential lethal impacts to 7,526 green sea turtles over the 30-year life of the projects would reduce the number of NA DPS green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal impacts would also result in a reduction in future reproduction, assuming some of the lost individuals would be female and would have survived otherwise to reproduce. For example, an adult green sea turtle will generally lay 3-4 clutches of eggs every 2-4 years, with a mean clutch size of 110-115 eggs per nest. The loss of sexually mature females could preclude the production of thousands of eggs and hatchlings, of which a small fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their potential contribution to future generations, and result in a reduction in NA DPS green sea turtle reproduction.

The NA DPS is the largest and most diverse of the 11 green sea turtle DPSs, with an estimated nester abundance of over 167,000 adult females from 73 nesting sites in multiple countries. Many of these nesting sites have high levels of abundance (<1000 nesters), including locations in Costa Rica, Cuba, Mexico, and Florida. All major nesting populations are experiencing long-term increases in abundance (Seminoff et al. 2015b). However, there are no significant nesting sites in the area of the northwestern Gulf of Mexico where projects' effects are likely to occur, and this area is used primarily as foraging habitat by NA DPS green sea turtles. The loss of up to 7,405 small juvenile and 121 (non-nesting) adult NA DPS green sea turtles from the Gulf of Mexico over a 30-year period would not have an appreciable effect on the overall distribution of this wide ranging DPS.

Nesting data described in Section 3 indicate at least 167,000 nesting females throughout the NA DPS. Based on this estimate, and using egg and hatchling survival rates described in Crouse et al. (1987), we calculate that the loss of 7,405 small juvenile green sea turtles would equal approximately 0.08% of a single year's juvenile production for the NA DPS. The one-time loss of 0.08% of a single year's juvenile production is not expected to have any appreciable effect on the survival of the NA DPS of green sea turtles.

167,000 nests[9] * 110[10] eggs/nest = 18,370,000 eggs
18,370,000 eggs * 0.675 survival rate to hatch = 12,399,750 hatchlings
12,399,750 hatchlings * 0.786 survival to 1 year = 9,746,204 juveniles produced
7,405 juveniles killed/9,746,204 juveniles produced = 0.08%

Whether the above described reductions in numbers and reproduction of this species will appreciably reduce the species likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends. In the Status of Species (Section 3.2.2), we presented the status of the NA 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 (Section 4.1.2), 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 NA DPS. In the Cumulative Effects (Section 6), we discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area.

In Section 3, we summarized the available information on the number of green sea turtle nesters and nesting trends at NA DPS beaches; all major nesting populations demonstrate long-term increases in abundance (Seminoff et al. 2015a). Therefore, nesting at the primary nesting beaches has been increasing over the course of the decades, against the background of the past and ongoing human and natural factors that have contributed to the Status of the Species (including the DWH oil spill). We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. In the absence of any total population estimates, nesting trends are the best proxy for estimating population changes. Since the nesting abundance trend information for the NA DPS of green sea turtle is increasing, we believe the combined potential lethal take of up to 7,405 small juvenile and 121 adult green sea turtles from the NA DPS over a 30-year period 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 the proposed actions are not expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle NA DPS in the wild.

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

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[9] Nesting females average approximately 3 nests per season, and mature females return to nest approximately once every 3 years, so an average of 1 nest per mature female in a population is a reasonable annual estimate.
[10] From Johnson, S. A., and L. M. Ehrhart. 1996. Reproductive ecology of the Florida green turtle: Clutch frequency. Journal of Herpetology 30(3):407-410.

- 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 1989-2019, 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 18), and indicate 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 loss of 7,405 small juvenile and 121 adult green sea turtles from the NA DPS, as a result of a worst case discharge from the proposed DWPs at some point over the next 30 years would result in a reduction in numbers; 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. Furthermore, the non-lethal effects would not affect the adult female nesting population or number of nests per nesting season. Thus, the proposed actions will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of NA DPS green sea turtles' recovery in the wild.

Conclusion
The combined potential lethal and non-lethal effects of the proposed actions over a 30-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NA DPS of green sea turtle in the wild.

Green Sea Turtle SA DPS
For this assessment, we assume the effects of a Worst Credible Oil Spill Discharge from the proposed actions would result in 14% of the exposures and mortalities of green sea turtles estimated for the DWH spill. The DWH PDARP assessment did not differentiate between the NA and SA DPSs of green sea turtles, but as explained above we assume that 4% of the green sea turtle impacts estimated in the DWH PDARP were SA DPS green sea turtles. The DWH PDARP estimated the number of oceanic juvenile green sea turtles exposed to, and killed by, the DWH spill and spill response activities, but did not identify any neritic (near-shore, age 3+) juveniles or adults that might have been effected. For this estimate, we conservatively assume that all of the unidentified neritic juveniles and adults (Table 17 and Table 18) were green sea turtles. We summarize the effects of a Worst Credible Oil Spill Discharge and oil spill response activities resulting from the proposed actions on the green sea turtle SA DPS by type of effect (exposure and mortality) and life stage (neritic age 3+ and oceanic juvenile) in Table 20 below.

Table 20. Summary of the Effects of a Worst Credible Oil Spill Discharge on SA DPS of Green Sea Turtles

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead

SA Green Oceanic Juvenile	829	86	223	309
SA Green Age 3+	33	4	1	5
Total	862	90	224	314

In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of SA DPS green sea turtles, we analyze how the action may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

We estimate that the proposed actions could expose up to 862 green sea turtles from the SA DPS to oil and oil spill response activities over the 30-year life of the projects. Of those 862 turtles, we estimate that 314 of them will suffer mortality, and that the remaining 548 will experience temporary effects and fully recover from these effects. These non-lethal exposures are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individuals suffering non-lethal injuries or stresses are expected to fully recover such that effects to these individuals would not result in reductions in reproduction or numbers of SA DPS green sea turtles. Any exposed turtles would be expected to stay within the same general area and no change in the distribution of SA DPS green sea turtles would be anticipated as a result of the projected non-lethal exposures.

The potential lethal impacts 314 green sea turtles over the 30-year life of the projects would reduce the number of SA DPS green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal impacts would also result in a reduction in future reproduction, assuming some of the lost individuals would be female and would have survived otherwise to reproduce. For example, an adult green sea turtle will generally lay 3-4 clutches of eggs every 2-4 years, with a mean clutch size of 110-115 eggs per nest. The loss of sexually mature females could preclude the production of thousands of eggs and hatchlings, of which a small fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their potential contribution to future generations, and result in a reduction in SA DPS green sea turtle reproduction.

It is unlikely that the lethal and non-lethal impacts expected over the 30-year life of the projects will have any measurable impact on the overall distribution of the SA DPS of green sea turtles. The in-water range of this DPS is extremely widespread, including significant nesting and feeding grounds along the West Coast of Africa (e.g. Guinea, Congo and Angola), as well as the Caribbean and the East Coast of South America (e.g. Venezuela, Suriname and Brazil). While no nesting occurs as far south as Uruguay and Argentina, both have important foraging grounds for the SA DPS. There are no nesting sites in the northwestern Gulf of Mexico where projects' effects are likely to occur, and this area is used exclusively as foraging habitat by SA DPS green sea turtles. Given the extremely broad distribution and the robust nesting populations throughout its range (conservatively estimated at over 63,000 nesters; Seminoff et al. 2015), the loss of up to 309 small juvenile and 5 (non-nesting) adult SA DPS green sea turtles from the Gulf of Mexico over a 30-year period would not have an appreciable effect on the overall distribution of this wide ranging DPS.

Nesting data described in Section 3 indicate at least 63,000 nesting females throughout the SA DPS. Based on this estimate, and using egg and hatchling survival rates described in Crouse et al. (1987), we calculate that the loss of 309 small juvenile green sea turtles would equal approximately 0.0084% of a single year's juvenile production for the SA DPS. The one-time loss of 0.0084% of a single year's juvenile production is not expected to have any appreciable effect on the survival of the SA DPS of green sea turtles.

63,000 nests[11] * 110[12] eggs/nest = 6,930,000 eggs
6,930,000 eggs * 0.675 survival rate to hatch = 4,677,750 hatchlings
4,677,750 hatchlings * 0.786 survival to 1 year = 3,676,712 juveniles produced
309 juveniles killed/3,676,712 juveniles produced = 0.0084%

Whether the above described 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 3.2.2), we presented the status of the SA 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 (Section 4.1.2), 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 SA DPS. In the Cumulative Effects (Section 6), we discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area.

Based on the robust population estimates (at least 63,000 nesters spread across 51 nesting sites) and stable or increasing trends in nesting observed over recent decades, against the background of the past and ongoing human and natural factors that have contributed to the Status of the Species (including the DWH oil spill), we believe these nesting trends are indicative of a stable or increasing population with a high number of sexually mature individuals. Given these conclusions, we believe the combined potential lethal take of up to 309 small juvenile and 5 adult green sea turtles from the SA DPS over a 30-year period will not have any measurable effect on the survival of this DPS. After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe that the proposed actions are not expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle SA DPS in the wild.

Recovery
The SA 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 SA DPS all occur in the Atlantic Ocean and

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[11] Nesting females average approximately 3 nests per season, and mature females return to nest approximately once every 3 years, so an average of 1 nest per mature female in a population is a reasonable annual estimate.
[12] From Johnson, S. A., and L. M. Ehrhart. 1996. Reproductive ecology of the Florida green turtle: Clutch frequency. Journal of Herpetology 30(3):407-410.

would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide until a new plan, specific to the SA DPS, is developed. 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.

There is no nesting of the SA DPS in Florida, but given the estimated nesting abundance of over 63,000 adult females in the South Atlantic, and the fact that all major nesting populations for which we have consistent monitoring data are experiencing long-term stability or increases in abundance (Seminoff et al. 2015b), the potential loss of 309 small juvenile and 5 adult green sea turtles from the SA DPS are not expected to result in any detectable reduction in the average annual nesting levels. We have no data on the total number of individuals in the SA DPS foraging grounds, but the documented stability and potential increases in adult females within the DPS is a good indication that the other sectors of the population are also stable or growing. Therefore, we believe the proposed actions are not expected to impede achieving the basic recovery objectives and are not expected to result in an appreciable reduction in the likelihood of the SA DPS of green sea turtles' recovery in the wild.

Conclusion
The combined potential lethal and non-lethal effects of the proposed actions over a 30-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of green sea turtle in the wild.

7.1.2.2 Kemp's Ridley Sea Turtle
The Kemp's ridley sea turtle was listed as endangered on December 2, 1970, under the Endangered Species Conservation Act of 1969, a precursor to the ESA. For this assessment, we assume the effects of a Worst Credible Oil Spill Discharge from the proposed actions would result in 14% of the exposures and mortalities of Kemp's ridley sea turtles estimated for the DWH spill. We summarize the effects of a potential Worst Credible Oil Spill Discharge related to the proposed actions on Kemp's ridley sea turtles by type of effect (exposure and mortality) and life stage (neritic age 3+ and oceanic juvenile) in Table 21 below.

Table 21. Summary of the Effects of a Worst Credible Oil Spill Discharge on Kemp's Ridley Sea Turtles

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Kemp's ridley juvenile	28,840	4,970	7,140	12,110
Kemp's ridley age 3+	3,080	294	138	434
Total	31,920	5,264	7,278	12,544

In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of Kemp's ridley sea turtles, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

We estimate that the proposed actions are could expose up to 31,920 Kemp's ridley sea turtles to oil and oil spill response activities over the 30-year life of the projects. Of those 31,920 turtles, we estimate that 12,544 of them will suffer mortality, and that the remaining 19,376 will experience temporary effects and fully recover from these effects. These non-lethal exposures are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individuals suffering non-lethal injuries or stresses are expected to fully recover such that effects to these individuals would not result in reductions in reproduction or numbers of Kemp's ridley sea turtles. Any exposed turtles would be expected to stay within the same general area and no change in the distribution of Kemp's ridley sea turtles would be anticipated as a result of the projected non-lethal exposures.

The potential loss of 12,544 individuals over the 30-year life of the projects would reduce the number of Kemp's ridley sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal impacts would also result in a reduction in future reproduction, assuming some of the lost individuals would be female and would have survived otherwise to reproduce. For example, an adult Kemp's ridley sea turtle will lay an average of 2.5 clutches of eggs every 2 years, with a mean clutch size of 100 eggs per nest. The loss of sexually mature females could preclude the production of thousands of eggs and hatchlings, of which a small fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their potential contribution to future generations, and result in a reduction in Kemp's ridley sea turtle reproduction.

It is unlikely that the lethal and non-lethal impacts expected over the 30-year life of the proposed actions will have any measurable impact on the overall distribution of Kemp's ridley sea turtles. While the primary range of Kemp's ridley sea turtles is within the Gulf of Mexico basin, less than 1 percent nest within the action area (primarily on North and South Padre Islands, which have only a 5% chance of being oiled in the Worst Credible Oil Spill Discharge scenario; Figure 38). Currently, over 90% of nesting occurs within a 76-mile stretch of beaches along the southwestern Gulf coast of Mexico (Gladys Porter Zoo 2017). Adult and juvenile Kemp's ridley sea turtles have also been documented in coastal and offshore waters of the U.S. Atlantic Ocean, and the species nesting distribution appears to be expanding in recent years with nesting recorded from beaches in Texas, Florida, Georgia, and the Carolinas. Nesting in these new areas has continued to increase in recent years (with oscillations mirroring those seen at the primary nesting beaches in Mexico), and in 2012, the first Kemp's ridley sea turtle nest was recorded in Virginia. We therefore conclude that the effects of the proposed actions are not expected to reduce the overall distribution of Kemp's ridley sea turtles.

Nesting data described in Section 3 indicate that the primary nesting sites in Mexico have seen significant fluctuations in nest numbers over the last decade, but the average number of nests over that period has been approximately 17,100 nests. Based on this estimate, and using egg and hatchling survival rates described in Crouse et al. (1987), we calculate that the loss of 12,110 small juvenile Kemp's ridley sea turtles would equal approximately 1.33% of a single year's juvenile production for the species. The one-time loss of 1.33% of a single year's juvenile production is not expected to have any appreciable effect on the survival of Kemp's ridley sea turtles.

17,100 nests * 100 eggs/nest = 1,711,650 eggs
1,711,650 eggs * 0.675 survival rate to hatch = 1,155,364 hatchlings
1,155,364 hatchlings * 0.786 survival to 1 year = 908,116 juveniles produced
12,110 juveniles killed/908,116 juveniles produced = 1.33%

Whether the above described 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 3.2.3), we presented the status of Kemp's ridley sea turtles, outlined threats, and discussed information on estimates of the number of nests and nesting trends at primary nesting beaches. In the Environmental Baseline (Section 4.1.2), 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 Kemp's ridley sea turtles. In the Cumulative Effects (Section 6), we discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area.

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 trajectory in Kemp's ridleys. Given the significant oscillations in nesting numbers over the past decade, it is impossible to predict how future trends may be affected. Nonetheless, data from 1990 to present continue to support that Kemp's ridley sea turtle are showing a generally increasing nesting trend. We believe this long-term increasing trend in nesting is evidence of an increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Given these conclusions, we believe the combined potential lethal take of up to 12,110 small juvenile and 434 adult Kemp's ridley sea turtles over a 30-year period will not have any measurable effect on the survival of this species. After analyzing the magnitude of the effects, in combination with the past, present, and future expected impacts to the species discussed in this Opinion, we believe that the proposed actions are not expected to cause an appreciable reduction in the likelihood of survival of Kemp's ridley sea turtles in the wild.

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 and the numbers from 2017 were very close to 25,000, indicating that the goal of 10,000 nesting females may have been reached (for 1 year only). The fact that these high nesting periods occurred following the devastating impacts of the DWH oil spill, along with the overall increasing long-term trend for the species, indicates that the species has maintained its ability to reach and exceed the stated recover goals. The long-term increase in Kemp's ridley sea turtle population is believed to be 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 1998; TEWG 2000). Based on the current population size and trends for this species, we believe that the potential reductions in numbers and reproduction that may be caused by the proposed actions will not have an appreciable effect on the recovery objective above, and are not likely to result in an appreciable reduction in the likelihood of Kemp's ridley sea turtles' recovery in the wild.

Conclusion
The combined potential lethal and non-lethal effects of the proposed actions over a 30-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of Kemp's ridley sea turtles in the wild.

7.1.2.3 NWA DPS of Loggerhead Sea Turtle

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 designating nine DPSs for loggerhead sea turtles on September 22, 2011. The NWA DPS, which is the only DPS represented in the action area, is listed as threatened. For this assessment, we assume the effects of a Worst Credible Oil Spill Discharge from the proposed actions would result in 14% of the exposures and mortalities of loggerhead sea turtles estimated for the DWH spill. We summarize the effects of a potential Worst Credible Oil Spill Discharge related to the proposed actions on the NWA DPS of loggerhead sea turtles by type of effect (exposure and mortality) and life stage (neritic age 3+ and oceanic juvenile) in Table 22 below.

Table 22. Summary of the Effects of a Worst Credible Oil Spill Discharge on NWA DPS of Loggerhead Sea Turtles

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Loggerhead Oceanic Juvenile	4,172	290	1,164	1,454
Loggerhead Age 3+	4,200	308	196	504
Total	8,372	598	1,360	1,958

In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of NWA DPS loggerhead sea turtles, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival
We estimate that the proposed actions could expose up to 8,372 loggerhead sea turtles to oil and oil spill response activities over the 30-year life of the projects. Of those 8,372 turtles, we estimate that 1,958 of them will suffer mortality, and that the remaining 6,414 will experience temporary effects and will fully recover from these effects. These non-lethal exposures are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individuals suffering non-lethal injuries or stresses are expected to fully recover such that effects to these individuals would not result in reductions in reproduction or numbers of loggerhead sea turtles. Any exposed turtles would be expected to stay within the same general area and no change in the distribution of loggerhead sea turtles would be anticipated as a result of the projected non-lethal exposures.

The potential loss of 1,958 individuals over the 30-year life of the projects would reduce the number of loggerhead sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal impacts would also result in a reduction in future reproduction, assuming some of the lost individuals would be female and would have survived otherwise to reproduce. For example, an adult female loggerhead sea turtle can lay approximately 4 clutches of eggs every 3-4 years, with 100-126 eggs per clutch. The loss of sexually mature females could preclude the production of thousands of eggs and hatchlings, of which a small fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their potential contribution to future generations, and result in a reduction in loggerhead sea turtle reproduction.

It is unlikely that the lethal and non-lethal impacts expected over the 30-year life of the projects will have any measurable impact on the overall distribution of the NWA DPS of loggerhead sea turtles. Aerial surveys indicate that NWA DPS loggerheads are distributed throughout U.S. waters as follows: 54% off the southeast U.S. coast, 29% off the northeast U.S. coast, 12% in the eastern Gulf of Mexico (outside of the action area) and only 5% in the western Gulf of Mexico, where the effects of the proposed actions may occur (TEWG 1998). There is no indication that these distributions were changed significantly by the impacts of the DWH oil spill (Dennis Klemm, NMFS Southeast Regional Sea Turtle Recovery Coordinator, pers. comm., August 4, 2020), and recent data shows that all of the nesting areas for the NWA DPS continue to be used by loggerheads, suggesting neither the in-water distribution nor the nesting distribution was changed as a result of the DWH spill in 2010. Based on this information we conclude that the effects of the proposed actions are not expected to reduce the overall distribution of the NWA DPS of loggerhead sea turtles.

Of the 5 recovery units delineated for the NWA DPS, the Northern Gulf of Mexico Recovery Unit (NGMRU; which nests on beaches from Franklin County, Florida, through Texas) is the unit most likely to be affected by the proposed actions. The percent of the overall population of the NWA DPS of loggerhead sea turtles represented by this recovery unit is relatively small compared to the Peninsular Florida Recovery Unit and the Northern Recovery Unit of the NWA DPS, and slightly larger than the other 2 recovery units. It is possible that some individuals from the other recovery units may be present in the northwest Gulf, and could affected by an oil spill from the proposed DWPs. However, we anticipate that the majority of impacts to loggerheads sea turtles will likely occur to turtles from the NGMRU because these are the only loggerhead sea turtles that nest in the areas that may be affected by the proposed actions.

Nesting data described in Section 3 indicate an average of 105,500 nests per year in the NWA DPS of loggerhead sea turtles. Based on this estimate, and using egg and hatchling survival rates described in Crouse et al. (1987), we calculate that the loss of 1,454 small juvenile loggerhead sea turtles would equal approximately 0.03% of a single year's juvenile production for the DPS. The one-time loss of 0.03% of a single year's juvenile production is not expected to have any appreciable effect on the survival of the NWA DPS of loggerhead sea turtles.

105,500[13] nests * 100[14] eggs/nest = 10,550,000 eggs
10,550,000 eggs * 0.675 survival rate to hatch = 7,121,250 hatchlings
7,121,250 hatchlings * 0.786 survival to 1 year = 5,597,303 juveniles produced
1,454 juveniles killed/5,597,303 juveniles produced = 0.03%

Whether the above described 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 absence of any total population estimates, nesting trends are the best proxy for estimating population changes. In Section 3.2.5, we summarized available information on the number of loggerhead sea turtle nesters and nesting trends. Nesting trends across all of the recovery units in the NWA DPS 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. These data indicate the NWA DPS population is large (i.e., several hundred thousand individuals), and that the abundance of neritic juvenile loggerheads is steady or increasing.

Based on the robust population estimates and strong upward trends in nesting observed over the past decade, and the minor potential effects on the overall juvenile population described above, we believe the combined potential lethal take of up to 1,454 small juvenile and 504 adult loggerhead sea turtles over a 30-year period will not have any measurable effect on the survival of this DPS. In analyzing the effects of the proposed actions, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed actions are not likely to cause an appreciable reduction in the likelihood of survival of the loggerhead sea turtle NWA DPS in the wild.

Recovery

The loggerhead recovery plan defines the recovery goal as "…ensur[ing] that each recovery unit meets its Recovery Criteria alleviating threats to the species so that protection under the ESA is no longer necessary" (NMFS and USFWS 2008a). The recovery plan for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2008b) 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. This plan anticipates that, with full implementation, the NWA population (DPS) will

[13] Based on 2017 estimate of total nests for Florida and the East Coast combined.
[14] Based on the lower end of the estimate of 100-126 eggs per nest from Dodd Jr. (1988).

recover within 50-150 years, but notes that reaching recovery in only 50 years would require a rapid reversal of the then declining trends of the Northern, Peninsular Florida, and Northern Gulf of Mexico Recovery Units. The higher end (150 years) assumes that additional time will be needed for recovery actions to bring about population growth.

The recovery objectives most pertinent to the potential impacts of the proposed actions are Numbers 1 through 4, and 8 (listed below):

1. Ensure that the number of nests in each recovery unit are increasing and that this increase corresponds to an increase in the number of nesting females.

2. 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.

3. Manage sufficient nesting beach habitat to ensure successful nesting.

4. Manage sufficient feeding, migratory, and internesting marine habitats to ensure successful growth and reproduction.

8. Recognize and respond to mass/unusual mortality or disease events appropriately.

Recovery Objectives 1 and 2, are the plan's overarching objectives and have associated specific demographic criteria. Currently, none of the specific criteria are being met, but the plan acknowledges that it will take 50-150 years to achieve recovery. Recent status information provided in Section 3 above, indicates the current population consists of several hundred thousand individuals and is showing encouraging signs of stabilizing and possibly increasing. These nesting surveys and in-water surveys indicate that the DPS is represented by a broad range of age classes, supports genetic heterogeneity, and a large number of sexually mature individuals producing viable offspring ((NMFS-NEFSC 2011)). The fact that these improvements over the past decade occurred following the impacts of the DWH oil spill, along with the overall increasing long-term trend for the species, indicates that the species has maintained its ability to reach and exceed the stated recover goals. The long-term increase in loggerhead sea turtle population is believed to be 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 1998; TEWG 2000). Based on the current population size and trends for this DPS, we believe that the potential reductions in numbers and reproduction that may be caused by the proposed actions will not have an appreciable effect on the recovery objective above, and are not likely to result in an appreciable reduction in the likelihood of loggerhead sea turtles' recovery in the wild.

Conclusion

The combined potential lethal and non-lethal effects of the proposed actions over a 30-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the loggerhead sea turtle NWA DPS in the wild.

7.1.2.4 Hawksbill Sea Turtles

The hawksbill sea turtle was listed as endangered throughout its entire range on June 2, 1970 under the Endangered Species Conservation Act of 1969, a precursor to the ESA. The effects of the DWH oil spill are used to project the effects of a Worst Credible Oil Spill Discharge from the proposed actions. For this assessment, we assume the effects of a Worst Credible Oil Spill Discharge from the proposed DWPs would result in 14% of the exposures and mortalities of loggerhead sea turtles estimated for the DWH spill. There were relatively few hawksbill sea turtles thought to be impacted by the DWH oil spill and related response efforts. Only 29 hawksbills were observed throughout the response period, with 4 treated and released, and one documented mortality. Because the DWH PDARP did not include an estimate of the number of adult hawksbill sea turtles effected, we conservatively assume that all of the unidentified adults were hawksbill sea turtles. We summarize the potential effects of a worst case discharge related to the proposed actions on hawksbill sea turtles by type of effect (exposure and mortality) and life stage (neritic age 3+ and oceanic juvenile) in Table 23 below.

Table 23. Summary of the Effects of a Worst Credible Oil Spill Discharge on Hawksbill Sea Turtles

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Hawksbill Oceanic Juvenile	1,199	84	335	419
Hawksbill Age 3+	826	89	37	126
Total	2,025	173	372	545

In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of hawksbill sea turtles, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

We estimate that the proposed actions could expose up to 2,025 hawksbill sea turtles to oil and oil spill response activities over the 30-year life of the projects. Of those 2,025 turtles, we estimate that 545 of them will suffer mortality, and that the remaining 1,480 will experience temporary effects and fully recover from these effects. These non-lethal exposures are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individuals suffering non-lethal injuries or stresses are expected to fully recover such that effects to these individuals would not result in reductions in reproduction or numbers of hawksbill sea turtles. Any exposed turtles would be expected to stay within the same general area and no change in the distribution of hawksbill sea turtles would be anticipated as a result of the projected non-lethal exposures.

The potential loss of 545 hawksbill sea turtles over the 30-year life of the projects would reduce the number of hawksbill sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal impacts would also result in a reduction in future reproduction, assuming some of the lost individuals would be female and would have survived otherwise to reproduce. For example, an adult female hawksbill sea turtle can lay approximately 3 clutches of eggs every 3-4 years, with 140 eggs per clutch. The loss of sexually mature females could preclude the production of thousands of eggs and hatchlings, of which a small fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their potential contribution to future generations, and result in a reduction in hawksbill sea turtle reproduction.

It is unlikely that the estimated lethal and nonlethal impacts of the proposed actions on hawksbill sea turtles would have any measurable impact on the overall distribution of the species. Hawksbills are broadly distributed throughout tropical and subtropical regions around the world, with a 2007 estimate of 21,212 to 28,138 adult females nesting at 83 different sites throughout 10 ocean regions. There are no known hawksbill nesting sites within the action area, and any individuals that may be affected by the proposed actions would likely be a mix of turtles from nesting populations in the Caribbean, Florida, and Mexico. In the western Atlantic, adult and juvenile hawksbills are distributed throughout the Caribbean Sea, off the coasts of Florida and Texas in the continental United States, in the Greater and Lesser Antilles, and along the mainland of Central America south to Brazil. Genetic research has shown that hawksbills of multiple nesting origins commonly mix in foraging areas (Bowen and Witzell 1996). Given this extremely broad, circumtropical distribution, and the number of individuals expected to be adversely affected by the proposed actions, it is unlikely that these actions will have any appreciable effect on the overall distribution of hawksbill sea turtles.

In Section 3.2.6 (Status of Species), we provide information on the population dynamics of the species. Hawksbills are broadly distributed throughout tropical and subtropical regions around the world, with a 2007 estimate of 21,212 to 28,138 adult females nesting at 83 different sites throughout 10 ocean regions. Hawksbills nest in low densities on scattered small beaches throughout the Caribbean, with the majority of nesting occurring in Mexico and Cuba. In Mexico, about 2,800 adult females were estimated to nest in Campeche, Yucatán, and Quintana Roo in 2003 (Spotila 2004), and Lutz (2003) estimated the total number of adult hawksbills living in the Caribbean to be 27,000. The most recent data on population trends is from 2012, which shows nesting populations throughout the Caribbean and mainland Central America continued to grow following the DWH spill (NMFS and USFWS 2013b).

The potential loss of 419 small juveniles would equal approximately 0.027% of a single year's juvenile production, based on the information detailed in Section 3 above (and using egg and hatchling survival rates described in Crouse et al. (1987)).

21,212 nests[15] * 140[16] eggs/nest = 2,969,680 eggs
2,969,680 eggs * 0.675 survival rate to hatch = 2,004,534 hatchlings
2,004,534 hatchlings * 0.786 survival to 1 year = 1,575,564 juveniles produced
419 juveniles killed/1,575,564 juveniles produced = 0.027%

[15] Based on the 2007 lower estimate of 21,212 nesting females, assuming each female lays 3 nests once every 3 years.
[16] From USFWS hawksbill fact sheet (http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm).

Based on the robust population estimates (21,212 to 28,138 adult females nesting at 83 different sites throughout 10 ocean regions in 2007) and stable or increasing trends in nesting observed throughout the Caribbean and mainland Central America following the impacts of the DWH oil spill, and the relatively minor estimated lethal take of small juveniles described above, we believe the proposed actions are not likely to cause an appreciable reduction in the likelihood of survival of hawksbill sea turtles in the wild.

Recovery
The Services' recovery plan for hawksbill sea turtles (NMFS and USFWS 2008c) provides a detailed explanation of the goals and vision for recovery for this species. The recovery objectives most pertinent to the potential impacts of the proposed actions are recovery objectives numbers 1 and 3 (listed below):

(1) The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests on at least five index beaches, including Mona Island and Buck Island Reef National Monument.

(3) Numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, United States Virgin Islands (USVI), and Florida.

With respect to regional trends, nesting populations in the Atlantic, and especially in the Insular Caribbean and Western Caribbean Mainland (where many of the individuals that would be affected by the proposed actions would likely originate from) are generally doing better than those in the Indo-Pacific regions. For instance, 9 of the 10 sites that showed recent increases are located in the Caribbean. The latest status review of hawksbill sea turtles (NMFS 2013b) found statistically significant increases in nesting on at least 7 index beaches in the Caribbean and mainland Central America, including Mona Island and Buck Island Reef National Monument. There has not been sufficient monitoring of in-water foraging areas to determine whether numbers of adults, subadults, and juveniles are increasing in these areas. However, given the relatively low densities of hawksbills occurring in the action area, the small number of adult and neritic juvenile hawksbills estimated to be impacted by the proposed actions, and the evidence of recent increases in nesting trends in the area, we believe that the estimated lethal and nonlethal take that may result from the proposed actions is not likely to have an appreciable effect on the recovery objectives above, and is not likely to result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild.

Conclusion
The combined potential lethal and non-lethal effects of the proposed actions over a 30-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of hawksbill sea turtles in the wild.

7.1.2.5 Leatherback Sea Turtle
The leatherback sea turtle was listed as endangered throughout its entire range on June 2, 1970, under the Endangered Species Conservation Act of 1969, a precursor to the ESA. For this assessment, we assume the effects of a Worst Credible Oil Spill Discharge from the proposed

actions would result in 14% of the exposures and mortalities of leatherback sea turtles estimated for the DWH spill. There were relatively few leatherback sea turtles thought to be impacted by the DWH oil spill and related response efforts. Only 21 leatherback sea turtles were observed alive in water from surface vessels and 2 others were observed from the air. No leatherbacks were captured or taken to rehab facilities, and none were observed dead throughout the response period. Due to these low observation levels the DWH PDARP was not able to quantify the amount or type of injuries incurred by leatherbacks. Additionally, there were too few leatherbacks detected during surveys for the DWH PDARP to estimate abundance or age class of leatherbacks exposed to the effects of the spill. Leatherback sea turtles do not nest within the action area, and there have not been any oceanic juvenile leatherback sea turtles documented in the Gulf of Mexico.

Based on the information detailed above, we assume that no oceanic juvenile leatherback sea turtles are likely to be adversely affected by the potential impacts of the proposed actions. While no adult leatherback sea turtles were documented to be adversely affected by the DWH oil spill, in order to be conservative towards the species, we assume that all of the unidentified adults impacted by the DWH oil spill were leatherback sea turtles, and that 14% of that number of adult leatherback sea turtles may be impacted by a potential Worst Credible Oil Spill Discharge from the proposed DWPs. We summarize the potential effects of a Worst Credible Oil Spill Discharge related to the proposed actions on adult leatherback sea turtles by type of effect (exposure and mortality) in Table 24 below.

Table 24. Summary of the Effects of a Worst Credible Oil Spill on Leatherback Sea Turtles

Species	Total Exposed	Heavily Oiled, Dead	Non-heavily Oiled, Dead	Total Dead
Leatherbacks Age 3+	826	89	37	126

In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of leatherback sea turtles, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

We estimate that the proposed actions could expose up to 826 adult and large juvenile leatherback sea turtles to oil and oil spill response activities over the 30-year life of the projects. Of those 826 turtles, we estimate that 126 of them will suffer mortality, and that the remaining 700 will experience temporary effects and fully recover from these effects. These non-lethal exposures are not expected to have a measurable impact on the reproduction, numbers, or distribution of the species. The individuals suffering non-lethal injuries or stresses are expected to fully recover such that effects to these individuals would not result in reductions in reproduction or numbers of leatherback sea turtles. Any exposed turtles would be expected to stay within the same general area and no change in the distribution of hawksbill sea turtles would be anticipated as a result of the projected non-lethal exposures.

The potential loss of 126 individuals over the 30-year life of the projects would reduce the number of leatherback sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal impacts would also result in a reduction in future reproduction, assuming some of the lost individuals would be female and would have survived otherwise to reproduce. For example, an adult female leatherback sea turtle can lay 10 clutches of eggs every 2-4 years, with 100 eggs per clutch and females can have fertility spans as long as 25 years (Hughes 1996). The loss of sexually mature females could preclude the production of thousands of eggs and hatchlings, of which a small fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their potential contribution to future generations, and result in a reduction in leatherback sea turtle reproduction.

It is unlikely that the estimated lethal and nonlethal impacts of the proposed actions on leatherback sea turtles would have any measurable impact on the overall distribution of the species. Leatherbacks nesting is very broadly distributed throughout the tropics and sub-tropics and they forage throughout an even broader range, into higher-latitude sub-polar regions. Important nesting areas in the western Atlantic Ocean occur in Florida, United States; St. Croix, U.S. Virgin Islands; Puerto Rico; Costa Rica; Panama; Colombia; Trinidad and Tobago; Guyana; Suriname; French Guiana; and southern Brazil. In the eastern Atlantic Ocean, a globally significant nesting population is concentrated in Gabon on the west coast of Africa, with additional widely dispersed but fairly regular nesting between Mauritania in the north and Angola in the south. In the Indian Ocean, major nesting beaches occur in South Africa, Sri Lanka, Andaman and Nicobar islands, with smaller populations in Mozambique, Java, and Malaysia. In the western Pacific Ocean, the main nesting beaches occur in the Solomon Islands, Papua Barat Indonesia, and Papua New Guinea, with additional nesting occurring in Vanuatu, Fiji, and southeastern Australia. In the eastern Pacific Ocean, important nesting beaches occur in Mexico and Costa Rica with scattered nesting along the Central American coast. Given this extremely broad, circumtropical distribution, the lack of nesting in the action area, and the relatively small number of individuals estimated to be adversely affected by the proposed actions, it is extremely unlikely that the proposed actions will have any appreciable effect on the overall distribution of leatherback sea turtles.

In Section 3.2.4 (Status of Species), we provide information on the population dynamics of the species. Leatherbacks are widespread throughout tropical, subtropical and temperate regions around the world. Spotila et al. (1996) estimated that the adult female leatherback population for just the Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, was about 27,600 (considering both nesting and internesting females), with an estimated range of 20,082-35,133. This estimate is consistent with the 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) estimated more recently by the Turtle Expert Working Group (TEWG;TEWG (2007). The TEWG (2007) also documented positive growth within major nesting areas of the western Atlantic, including an estimated three-generation abundance change of +3%, +20,800%, +1,778%, and +6% in Trinidad, Guyana, Suriname, and French Guiana, respectively. However, subsequent analysis using data up through 2017 has shown decreases in this stock, with an annual geometric mean decline of

10.43% over what they described as the short term (2008-2017) and a long-term (1990-2017) annual geometric mean decline of 5% (Northwest Atlantic Leatherback Working Group 2018).

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

In Florida, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005 and Tiwari et al. (2013) reported an annual growth rate of 9.7% and a three-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, followed by a partial rebound in 2018. The annual geometric mean trend for Florida has been a decline of almost 7% from 2008-2017, but the long-term trend (1990-2017) remains positive with an annual geometric mean increase of over 9% (Northwest Atlantic Leatherback Working Group 2018).

Based on the robust population estimates (34,000-95,000 total adults) and despite the recent declines in nesting at the closest nesting beaches (in Florida and the Northern Caribbean), we believe that the small amount of estimated lethal take of adults and large juveniles described above (with no expected effects to the distribution of the species) is not likely to cause an appreciable reduction in the likelihood of survival of leatherback sea turtles in the wild.

Recovery

The Services' recovery plan for leatherback sea turtles in the U.S. Caribbean, Atlantic and Gulf of Mexico (NMFS and USFWS 2008c) provides a detailed explanation of the goals and vision for recovery for this species. The recovery objective most pertinent to the impacts that may be caused by the proposed actions is Number 1:

1) The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico, St. Croix, USVI, and along the east coast of Florida.

We believe the proposed actions are not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. As discussed above, the most recent analysis from 2018 has shown a reverse in trends, as the Culebra, St. Croix, and Florida nesting populations have decreased in recent years, although the long-term trend in Florida remains positive. It is unclear whether these declines represent a shift in nesting locations, changes in reproductive output, actual declines in the adult female population, or some combination of those factors. However, the small amount of estimated lethal take of adults and large juveniles described above (126 individuals) is not likely to have any appreciable effect on the overall nesting trends in the Northwest Atlantic, and is not likely to impede the progress toward achieving this recovery objective. We therefore believe the proposed actions will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild.

Conclusion
The combined potential lethal and non-lethal effects of the proposed actions over a 30-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of leatherback sea turtles in the wild.

7.1.3 ESA-Listed Fish

The two ESA-listed fish species likely to be found within the action area are the giant manta ray and the oceanic whitetip shark. These two species differ in morphology, physiology, behavior, and ecology, but both are sparsely populated within the action area, and both are expected to be exposed to the same stressors from the proposed actions. In this section, we first summarize the stressors to which giant manta rays and the oceanic whitetip sharks will be exposed. Following this, we detail our integration and synthesis for each species in which we rely on and summarize information presented in the *Effects of the Action on Species,* the *Species Status*, the *Environmental Baseline*, and the *Cumulative Effects* Sections presented above.

The only stressor we believe is likely to adversely affect giant manta rays and oceanic whitetip sharks is that of exposure to oil and dispersants resulting from an oil spill and the oil spill response activities, which are expected to reduce the fitness of individuals exposed and possibly result in mortality depending on the severity of exposure.

7.1.3.1 Oceanic whitetip shark

The oceanic whitetip shark was listed as threatened under the ESA in 2018. This pelagic species is distributed worldwide in tropical and subtropical waters. While there is no range-wide abundance estimate available, it is thought to have once been one of the most abundant sharks in the ocean. Catch data from individual ocean basins indicate that the populations have undergone significant declines (Young et al. 2017). In the northwestern Atlantic and GoM, the oceanic whitetip shark was described historically as widespread, abundant, and the most common pelagic shark in warm waters. Recent information, however, suggests the species is now relatively rare in this region, with declines estimated to be between 57% and 88% (Young et al. 2017). While little information on genetic diversity exists for the species, some data indicate they have low genetic diversity making the species susceptible to inbreeding and 'Allee' effects, although the extent to which is currently unknown. There is mixed evidence regarding genetic structuring and population differentiation across ocean basins, but to date there is no unequivocal evidence for genetic discontinuity or marked separation between Atlantic and Indo-Pacific subpopulations (Young et al. 2017).

In the *Status of Species* and *Environmental Baseline* Sections, we identified fisheries interactions, from both targeted and non-targeted (i.e., bycatch) fisheries, as the main threat to the species. Due to the species vertical and horizontal distribution, oceanic whitetip sharks are frequently caught as bycatch in many commercial fisheries, including pelagic longline fisheries targeting tuna and swordfish, as well as purse seine, gillnet, and artisanal fisheries. In addition, they are targeted by some fisheries for their large, morphologically distinct fins, which sell for a high price in the Asian fin market. Given the inadequacy of existing regulatory measures to manage these fisheries at a global scale, fisheries interactions are expected to remain a threat to the species as a cumulative effect for the foreseeable future.

Oceanic whitetip sharks are free-swimming, often in deeper, pelagic waters and may aspirate spilled oil and dispersants in the water column through their gill filaments. Some small number of oceanic whitetip sharks are likely to be exposed to oil, and those exposures would likely result in effects similar to other marine species, including fitness reduction and possibly leading to mortality. Because there are no abundance estimates for oceanic whitetip sharks in the GoM, and there were no analyses of impacts to whitetip sharks in the DWH PDARP, we are not able to quantify an estimated number individuals expected to be exposed to oil or the oil spill response activities, or the number of individuals expected to suffer mortality as a result of such exposure for this species. Instead we use the spatial area estimated to be affected by the worst case discharge scenario for the SPOT DWP as an ecological surrogate for the amount and extent of potential impacts to oceanic whitetip sharks. In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of oceanic whitetip sharks, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

Oil and dispersants could contact the species' skin, potentially having adverse consequences depending on the severity of exposure. Finally, oceanic whitetip sharks could also ingest oil and dispersants, or both if their prey become contaminated, and oil and dispersants could also affect prey availability more generally. In our *Effects Analysis*, we estimated that 47,000 km^2 is likely to be impacted by a worse case discharge and oil spill response activities from the DWP projects. We are unable to quantitatively estimate the number of oceanic whitetip sharks likely to be adversely affected due to exposure to oil and dispersants within a 47,000 km^2 area in the western Gulf of Mexico due to the lack of abundance information within the action area. However, given that the majority of the action area (area that may be affected by a Worst Credible Oil Spill Discharge) has shallow water depths where oceanic whitetip sharks are not expected to occur, and the species abundance is generally low even within the deeper sections of the action area, we expect that within 47,000 km^2 in the western Gulf of Mexico, a small number of individuals are likely to be exposed to oil and dispersants at levels that would impact fitness, with even fewer expected to be exposed to levels that would result in mortality. Given the overall low exposure to oil spills and dispersants at levels that would impact individual fitness, we do not anticipate population-level effects to the greater Atlantic subpopulation. Therefore, we do not expect that the potential impacts that could result from the proposed actions will reduce appreciably the likelihood of survival of the oceanic whitetip shark in the wild.

Recovery

NMFS has not yet completed a recovery plan for oceanic whitetip sharks, but has developed a recovery outline, which identifies ongoing and significantly high rates of fishing mortality driven by incidental bycatch, particularly of juveniles, as the most significant and ongoing threat to the species. (https://media.fisheries.noaa.gov/dam-migration/final_oceanic_whitetip_recovery_outline.pdf). The initial focus of the interim recovery strategy is:

(1) to stabilize population trends through reduction of threats, such that the species is no longer declining throughout a majority of its range; and

(2) to gather additional information through research and monitoring on the species' current distribution and abundance; reproductive periodicity and seasonality; location of breeding and nursery grounds; and mortality rates in commercial fisheries (including at-vessel and post-release mortality).

We believe the proposed actions are not likely to impede the interim recovery strategy above and will not result in an appreciable reduction in the likelihood of oceanic whitetip sharks' recovery in the wild. The proposed actions will have no effect on the primary threat to this species (i.e., incidental bycatch), and a Worst Credible Oil Spill Discharge will likely affect a relatively small number of individuals, and thus is not likely to impede the progress toward achieving this recovery strategy. We therefore believe the proposed actions will not reduce appreciably the likelihood of recovery of the oceanic whitetip shark in the wild.

7.1.3.2 Giant manta ray

Like oceanic whitetip sharks, the giant manta ray was recently (2018) listed as threatened under the ESA. It occupies tropical, subtropical, and temperate oceanic waters and productive coastlines throughout the world. They are commonly found offshore in oceanic waters, but sometimes in shallow waters as well (Lawson et al. 2017; Miller and Klimovich 2017). There are at least 11 identified subpopulations with population size estimates ranging from 100 to 1,500 individuals based on anecdotal diver or fisherman observations (FAO 2012; Miller and Klimovich 2017). Abundance data from the Flower Garden Banks Marine Sanctuary in the GoM provides an estimate of more than 70 individuals (Miller and Klimovich 2017). While data on global trends of the species are unavailable, in the Indo-Pacific there have been decreases in landings of up to 95% (Miller and Klimovich 2017). The species is considered highly migratory, and thus genetically well-connected, but tagging, stable isotope, and genetic data from the Pacific Ocean off the coast of Mexico suggest population structuring between offshore and coastal giant manta rays (Stewart et al. 2016a). In addition, some have suggested there may be a subspecies of giant manta ray resident to the Yucatán (Hinojosa-Alvarez et al. 2016). However, the best available data do not indicate genetic discreteness between giant manta rays in the Atlantic and those in the Indo-Pacific and eastern Pacific (Miller and Klimovich 2017).

As discussed in the *Status of Species* and *Environmental Baseline* Sections, interactions with commercial fisheries are the main threat to the species. Along with other mobulids, giant manta rays are targeted for their gill rakers, which are dried and sold in Asia (O'Malley et al. 2017). Based on the doubling of the amount of mobulid gill rakers in Asian markets from 2011 to 2015, we expect targeted commercial fishing to remain a threat to the species as a cumulative effect for the foreseeable future. In addition to being targeted for their gill rakers, giant manta rays are also bycaught in industrial purse seine and artisanal gillnet fisheries, particularly in the eastern Pacific and the Indo-Pacific (Miller and Klimovich 2017).

A small number of giant manta rays are likely to be exposed to oil and response activity impacts, and those exposures would likely result in effects similar to other marine species including fitness reduction and possibly leading to mortality. Because there are no abundance estimates for giant manta rays for the GoM beyond the 70 individuals documented at FGBNMS, and there were no analyses of impacts to giant manta in the DWH PDARP, we are not able to quantify an estimated number of exposures or mortalities for this species. In determining whether the effects of the proposed actions may result in an appreciable reduction in the likelihood of survival and recovery of giant manta rays, we analyze how the actions may affect the numbers, reproduction, or distribution of the species, and the effect of any reduction in numbers, reproduction, or distribution on the likelihood of survival and recovery of the species. We address survival first, below.

Survival

Effects to giant manta rays from exposure to oil and dispersants are similar to those previously described for oceanic whitetip sharks. These include aspiration of oil/dispersants, contact between oil/dispersants and an individual's skin, ingestion of oil/dispersants through the ingestion of contaminated prey, and direct affects to prey availability. In our *Effects Analysis*, we estimated that 47,000 km^2 is likely to be impacted by a worse case discharge and oil spill response activities from the DWP projects. Given the lack of abundance information for most of the action area (except in the Flower Garden Banks where oil has a very low probability of reaching, even in the event of a worst case discharge), we are unable to quantitatively estimate the number of giant manta rays likely to be adversely affected due to exposure to oil and dispersants within a 47,000 km^2 area in the western Gulf of Mexico. However, given that data indicate the species abundance is generally low within the action area, we expect that within 47,000 km^2 in the western Gulf of Mexico only a small number of individuals are likely to be exposed to oil and dispersants at levels that would impact fitness, with even fewer expected to be exposed to levels that would result in mortality.

As mentioned above, there is some evidence of population differentiation in certain areas, and some have even suggested a subspecies of giant manta rays exists off the coast of Yucatán peninsula (Hinojosa-Alvarez et al. 2016). However, currently, giant manta rays in the Atlantic, Indo-Pacific, and eastern Pacific are all considered to part of the same genetic population (Miller and Klimovich 2017). Thus, we consider population-level effects to be those effects to the global population/species level. Based on the estimated overall low exposure of giant manta rays to oil spills and dispersants at levels that would impact individual fitness, we do not expected effects at the global population, species level. Therefore, we do not expect that the potential effects of the proposed actions will reduce appreciably the likelihood of survival of the giant manta ray in the wild.

Recovery

NMFS has not yet completed a recovery plan for giant manta rays, but has developed a recovery outline, which identifies overutilization for commercial purposes, as the most significant threat to the species. (https://media.fisheries.noaa.gov/dam-migration/giant_manta_ray_recovery_outline.pdf). The initial focus of the interim recovery strategy is:

(1) to stabilize population trends through reduction of threats, such that the species is no longer declining throughout a majority of its range; and

(2) to gather additional information through research and monitoring on the species' current distribution and abundance; reproductive periodicity and seasonality; location of breeding and nursery grounds; and mortality rates in commercial fisheries (including at-vessel and post-release mortality).

We believe the proposed actions are not likely to impede the interim recovery strategy above and will not result in an appreciable reduction in the likelihood of giant manta rays' recovery in the wild. The proposed actions will have no effect on the primary threat to this species (i.e., overutilization for commercial purposes), and a Worst Credible Oil Spill Discharge will likely affect a relatively small number of individuals, and thus is not likely to impede the progress toward achieving this recovery strategy. We therefore believe the proposed actions will not reduce appreciably the likelihood of recovery of the giant manta ray in the wild.

7.1.4 Species Integration and Synthesis Conclusions

Table 25 below summarizes conclusions for ESA-listed species determined to be adversely affected by the proposed actions. The proposed actions are not likely to jeopardize the continued existence of any ESA-listed species.

Table 25. Summary of Integration and Synthesis for ESA-Listed Species Likely to Be Adversely Affected

Species	Total Exposures to Oil and Oil Spill Response Activities	Mortality Resulting from Exposure	Jeopardy
Sperm Whale	37	13	No
Green Sea Turtle NA DPS Sea Turtle	20,684	7,526	No
Green Sea Turtle SA DPS Sea Turtle	862	314	No
Kemp's Ridley Sea Turtle	31,920	12,544	No
Loggerhead Sea Turtle	8,372	1,958	No
Hawksbill Sea Turtle	2,025	545	No
Leatherback Sea Turtle	826	126	No
Oceanic Whitetip Shark*	47,000 km^2	Not available	No
Giant Manta Ray*	47,000 km^2	Not available	No

* Ecologically relevant surrogate measured by total area of sea surface impacted

7.2 Critical Habitat Destruction/Adverse Modification Analysis

7.2.1 Loggerhead Sea Turtle Critical Habitat

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

This analysis takes into account the geographic and temporal scope of the proposed action, recognizing that "functionality" of critical habitat necessarily means that it must now and must continue in the future to support the conservation of the species and progress toward recovery. The analysis takes into account any changes in amount, distribution, or characteristics of the critical habitat that will be required over time to support the successful recovery of the species. Destruction or adverse modification does not depend strictly on the size or proportion of the area adversely affected, but rather on the role the action area and the affected critical habitat serves with regard to the function of the overall critical habitat designation, and how that role is affected by the action.

In the *Status of Species and Critical Habitat* and *Environmental Baseline* Sections, habitat loss or alteration is identified as one of the primary effects of climate change. Larger, more frequent storms threaten coastal and offshore habitats. Similar to the assessment for ESA-listed species, we consider global climate change, in addition to the other natural and anthropogenic stressors affecting critical habitats.

A portion of the NWA DPS of loggerhead sea turtles critical habit *Sargassum* LOGG-S-2 unit (*Sargassum* habitat) is found within the action area. Our effects analysis determined that potential oil spills and related oil spill response activities are likely to adversely affect *Sargassum* habitat. To conduct our destruction and adverse modification analysis, we consider the essential physical and biological features (PBFs) of loggerhead sea turtles critical habitat Unit LOGG-S-2 and evaluate the effects of the proposed actions on those essential features, both in the short-term and long-term.

Loggerhead sea turtle *Sargassum* habitat is described as developmental and foraging habitat for young loggerhead sea turtles where surface waters form accumulations of floating material, especially *Sargassum*. PBFs that support this habitat are:

- convergence zones, surface-water downwelling areas, the margins of major boundary currents (Gulf Stream), and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for the optimal growth of *Sargassum* and inhabitance of loggerhead sea turtles;
- *Sargassum* in concentrations that support adequate prey abundance and cover;

- available prey and other material associated with *Sargassum* habitat including, but not limited to, plants and cyanobacteria and animals native to the *Sargassum* community such as hydroids and copepods; and
- sufficient water depth and proximity to available currents to ensure offshore transport (out of the surf zone), and foraging and cover requirements by *Sargassum* for post-hatchling loggerhead sea turtles (i.e., greater than 10 meters depth).

Sargassum habitat is vulnerable to oil spills and spill response activities related to the proposed actions. Oil can be carried by currents into convergence zones where *Sargassum* is also accumulating. Physical processes, such as convergent currents and fronts that play a role in transporting, retaining, and concentrating *Sargassum*, are the same processes that act to concentrate oil, thus increasing the exposure of *Sargassum* associated organisms to oil. Indeed, *Sargassum* habitats could act as a natural boom to contain spilled oil. Oiled *Sargassum* would be removed from the environment (as part of any clean-up response activity) along with the associated prey community. Consequently, reductions in this habitat are likely with a large oil spill. The amount and breadth of the reduction depends on the location of the spill and is proportional to the size and the seasonal timing of the spill.

We estimate a Worst Credible Oil Spill Discharge from the proposed DWPs could impact roughly 47,000 km^2 of the GoM *Sargassum* habitat. Therefore, a Worst Credible Oil Spill Discharge could impact approximately 12% of the approximately 393,053.75 km^2 area designated as the GoM *Sargassum* habitat. By comparison, the DWH oil spill resulted in a the loss of approximately 23% of the *Sargassum* habitat in the northern GoM (at the time of the spill) due to direct exposure to DWH oil on the ocean surface (Trustees 2016). The loss of *Sargassum* habitat during DWH was likely exacerbated by the use of oil dispersants, in-situ burning, and surface skimming of oil/*Sargassum* mats (Powers et al. 2013). An extremely large oil spill (e.g., > 500,000 bbls), would likely result in widespread, sea-scape level impacts and could make the ability of juvenile turtles to locate suitable *Sargassum* habitat difficult, particularly if dispersants are used in the aftermath of such a spill.

Based on the best available information, we expect that a Worst Credible Oil Spill Discharge and oil spill response activities resulting from the proposed actions will adversely affect the following PBFs that provide adequate prey and cover for juvenile loggerhead sea turtles:

- concentrations of *Sargassum* habitat
- concentrations of available prey and other material associated with *Sargassum* habitat.

The effects of oil exposure on *Sargassum* habitat could be severe and last for days, weeks or even months. However, the ephemeral nature and annual cycle of rapid growth, movement, and subsequent senescence, allows *Sargassum* to repopulate in the GoM in the year subsequent to a very large oil spill. In the *Effects Section*, we assessed both the short-term and long-term effects of oil spills on the essential features of *Sargassum* habitat. We considered aspects of the algae's life cycle including seasonal movements and drift rate within the action area, growth rate, longevity, and resiliency to environmental disturbances. The amount of *Sargassum* exposed to an oil spill within the action area will depend to a large extent on the time of year the spill occurs, given the seasonality and cyclical movement of *Sargassum* in the northern GoM. Continuous exposure of a particular *Sargassum* patch to oil could last days, weeks, or months depending on the size and location of the spill and other factors (e.g., wind speed and direction, season, and type of oil). More heavily oiled patches that are closer to the spill source at the time of the spill, and areas exposed to both oil and oil dispersants, will likely die-off or sink to the ocean bottom.

While the adverse effects of an extremely large oil spill on *Sargassum* communities within a given annual life cycle (described above) are well documented, the longer-term impacts in subsequent years or decades are not known. Although approximately 23% of all *Sargassum* habitat in the northern GoM was heavily exposed to oil in the 2010 DWH spill, follow-up aerial surveys in 2011 and 2012 documented a four-fold increase in *Sargassum* abundance. These results suggest that *Sargassum* can repopulate in the GoM within a year or two of an extremely oil spill. These results also suggest that even in the case of an extremely large oil spill, the effects to designated critical habitat for the NWA DPS of loggerhead sea turtles are likely to temporally and spatially localized events.

Given its fast growth rate, continuous motion, and somewhat ephemeral nature, we expect a relatively high turnover rate for *Sargassum* patches under normal conditions, and we believe that *Sargassum* habitat lost due to an oil spill from the proposed projects will likely be replaced over time by the combination of movement by unexposed (or lightly exposed) existing patches and through new growth and that the adverse effects to from an oil spill and the response activities will not preclude the ability of *Sargassum* to grow new patches and counter those effects. Therefore, we determine the proposed actions will not appreciably diminish the value of loggerhead sea turtle designated critical habitat for the conservation of the species.

7.2.2 Critical Habitat Integration and Synthesis Conclusions

Table 26 summarizes conclusions for adverse modifications to the designated critical habitats for the NWA DPS of the loggerhead sea turtle. NMFS found that some PBFs of the *Sargassum* habitat would be adversely affected by the proposed action. However, we believe the effects would not rise to the level that would destroy or adversely modify this critical habitat. We determine the effects will not appreciably diminish the values of the designated critical habitat for the conservation of the species.

Table 26. Summary of Integration and Synthesis for Designated Critical Habitat based on review of effects to PBFs from oil spills.

Species Designated Critical Habitat	Area Impacted	Destruction or Adverse Modification
NWA DPS Loggerhead Sea Turtle	47,000 km^2	No

8 CONCLUSION

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

In addition, after reviewing the current status of loggerhead sea turtle critical habitat (LOGG-S-2), the Environmental Baseline, the Effects of the Proposed Actions, and the Cumulative Effects, it is our Opinion that the proposed action will not reduce the ability of designated critical habitat to support loggerhead sea turtle conservation. Given the nature of the proposed actions and the information provided above, we conclude that the action, as proposed, is likely to adversely affect, but is not likely to destroy or adversely modify, loggerhead sea turtle Critical Habitat unit LOGG-S-2.

9 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. *Take* is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. *Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the reasonable and prudent measures and the terms and conditions of the incidental take statement (ITS) of the Opinion. Incidental take statements serve a number of functions. In addition to providing an exemption from the prohibition on take in Section 9, an ITS identifies reasonable and prudent measures necessary or appropriate to minimize the impact of all incidental take on the species and monitors the action's effects on the species to assist in determining whether consultation should be reinitiated.

This ITS only covers take that is incidental to the proposed actions, as described in Section 3. Thus, it does not cover take incidental to potential future-planned actions, such as the approval of Facility Response Plans described in 33 CFR Part 154 subpart F or of oil spill response plans described in 30 CFR Part 254 subpart B, that are excluded from the proposed actions. The approval or implementation of such plans by a federal action agency may require a future ESA consultation.

The take of oceanic white tip shark and giant manta ray by the proposed actions is not prohibited, as NMFS has not promulgated Section 4(d) rules for these threatened species. 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 Cir. 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 or not reinitiation of consultation is required. It also ensures that we identify Reasonable and Prudent Measures we believe are necessary or appropriate to minimize the impact of such incidental take.

Section 7(b)(4)(C) of the ESA provides that take of ESA-listed marine mammals may be included in the ITS of a biological opinion only if the taking is authorized under §101(a)(5) of the MMPA. No incidental take of listed marine mammals has been authorized under Section 101(a)(5) of the MMPA. As explained below, no take of ESA-listed marine mammals is anticipated to occur as result of the construction, operation, and decommissioning phases of the proposed actions, including any project related activities or incidents, and no statement on incidental take of protected marine mammals is provided. Thus, no take of protected marine mammals is authorized under this ITS. Nevertheless, the action agencies must immediately notify (within 24 hrs, if communication is possible) NMFS' Office of Protected Resources if a take of a listed marine mammal occurs.

Furthermore, this ITS provides no exemption from the ESA's take prohibition for any ESA-listed species from oil spills associated with the proposed actions because oil spills are not a lawful activity under the Clean Water Act.[17] Nevertheless, if any take of ESA-listed species under NMFS's purview occurs as a result of an oil spill from either of the proposed actions, it shall be immediately reported to takereport.nmfsser@noaa.gov (include Opinion issue date, and the NMFS ECO tracking number SERO-2020-00075).

The USCG and MARAD have a continuing duty to regulate the activity covered by this incidental take statement. If the USCG and MARAD (1) fail to assume and implement the terms and conditions or (2) fail to require the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the USCG and MARAD must report to NMFS, any impact on the species specified in the Incidental Take Statement (50 CFR §402.14(i)(3)).

9.1 Anticipated Amount or Extent of Incidental Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent of such incidental taking on the species (50 C.F.R. §402.14(i)(1)(i)). As defined by 50 CFR § 402.02, "effects of the action" include consequences to the listed species or critical habitat that are caused by the proposed action and that are reasonably certain to occur. As explained in the Effects of the Action section, we anticipate the construction, operation, and decommissioning phases of the proposed actions, including any project related activities or incidents are not reasonably certain to result in any adverse effects to Kemp's ridley sea turtle, loggerhead sea turtle (NWA DPS), green sea turtle (NA and SA DPSs), leatherback sea turtle, giant manta ray, oceanic whitetip shark, or sperm whale. As a result, we concluded that the proposed actions may affect, but are not likely to adversely affect, these species and that incidental take is not likely to occur. Consequently, we specify that the amount of incidental take of these species associated with these elements of the

[17] The Clean Water Act (33 USC § 1251 et seq.), as amended by the Oil Pollution Act of 1990 (33 USC § 2701 et seq.), prohibits discharges of harmful quantities of oil, as defined at 40 CFR 110.3, into waters of the United States.

proposed actions is anticipated to be zero. If any take of ESA-listed species under NMFS's purview occurs during in-water construction or other activities authorized under this Opinion, it shall be immediately reported to takereport.nmfsser@noaa.gov (include Opinion issue date, and the NMFS ECO tracking number SERO-2020-00075). If there is any incidental take of ESA-listed species during construction, operation, or decommissioning phases of the proposed actions, we will consider the specified amount or extent of incidental take to be exceeded, and pursuant to 50 CFR §§ 402.14(i)(4) and 402.16, consultation must be reinitiated immediately.

While this ITS provides no exemptions for take of any ESA-listed species from oil spills associated with the proposed actions, we must clearly define the amount and extent of take that is anticipated and analyzed in this Opinion in order to know if and when that level of take might be exceeded, and reinitiation of consultation is necessary. We also must characterize and define this anticipated amount and extent of take in a manner that is easily measurable, so that we can clearly determine when that level has been reached. In the sections above, we developed an estimate of the numbers of ESA-listed species anticipated to be exposed to and killed by a potential Worst Credible Oil Spill Discharge, so that we could use those numbers to inform our jeopardy analysis for those species. However, during an actual oil spill event, it is very difficult to accurately quantify the numbers of individual animals that are exposed to and killed by the active oil spill. For example, it took years and extensive biological studies and research before we were able to quantify the amount and extent of take caused by the DWH oil spill. We must therefore choose an ecologically relevant surrogate, that is easily measured in real time, to use as a proxy indicator of the amount and extent of take caused by the proposed actions.

The analysis used to estimate the numbers of ESA-listed species that may be adversely affected by the proposed actions is based on the quantity of oil spilled. All other calculations that led us to numbers of individuals affected, originated from the base assumption that up to 687,602 bbl of oil could be spilled in a Worst Credible Oil Spill Discharge scenario. We therefore use this parameter to provide a measurable proxy for the amount and extent of take anticipated for the proposed actions over the 30-year life of the projects. To ensure that the anticipated amount and extent of take is not being exceeded throughout the 30-year life of the projects, we will keep track of any oil spills that result from the projects' operations, and calculate the total amount spilled every 5 years. If the total amount of oil spilled over a 5-year period exceeds 114,600 bbl (1/6 of the amount anticipated over the 30-year life of the project), reinitiation of consultation will be required in order to ensure the proposed projects are not likely to jeopardize the continued existence for ESA-listed species or destroy or adversely modify designated critical habitat.

9.2 Effect(s) of the Take

NMFS has determined that the anticipated take specified in this Opinion is not likely to jeopardize the continued existence of green, Kemp's ridley, leatherback, loggerhead, and hawksbill turtles; giant manta ray, oceanic whitetip shark, and sperm whale.

9.3 Reasonable and Prudent Measures (RPMs)

Section 7(b)(4) of the ESA requires NMFS to issue to any 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. It also states that RPMs necessary to minimize the impacts from the agency action, and terms and conditions to implement those measures, must be provided and followed. The RPMs and terms and conditions are required, per 50 CFR 402.14 (i)(1)(ii) and (iv), to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species.

We are not requiring any RPMs or terms and conditions to minimize the take of ESA-listed species associated with the construction, operation, and decommissioning phases of the proposed actions, including any project related activities or incidents, because we do not anticipate incidental take of these species. Therefore, there is no amount or extent of incidental take to minimize and no RPMs to minimize take of these species are necessary or appropriate.

Similarly, there are no RPMs or terms and conditions to minimize the take of ESA-listed species associated with oil spills resulting from the proposed actions. The oil and gas industry, along with federal and state regulatory agencies have been developing, testing, and implementing comprehensive safety protocols and regulations designed to prevent, minimize and mitigate oil spills from offshore facilities for many decades. Countless professional engineers and operational safety experts have focused their careers on the development of the comprehensive suite of safety regulations, protocols and equipment that are currently required for all offshore petroleum production and distribution facilities. Based on these facts, NMFS has not attempted to develop any new or novel measures that might improve on spill prevention and safety protocols and regulations that are currently in place. Instead, we rely on SPOT, TGL, and the federal regulatory action agencies to strictly adhere to all safety regulations, criteria and protocols that are currently in place to prevent, minimize, and mitigate any oil spills from the proposed projects.

We have considered whether there is sufficient monitoring in place to allow us to know if the specified amount or extent of incidental take has been exceeded. The USCG and MARAD have a continuing duty to regulate the activities covered by this ITS. To monitor the impact of the incidental take, the USCG and MARAD must report the progress of the actions and their impact on ESA-listed species to SERO PRD as specified in this ITS [50 CFR 402.14(i)(3)].

Because there are several elements of the proposed construction-related pile driving operations that are somewhat novel, there remains a level of uncertainty as to the actual level of sound propagation and acoustic impacts that may be experienced by ESA-listed species present in the project areas. For these reasons, it will be necessary for SPOT and TGL to conduct comprehensive acoustic monitoring during the proposed pile driving activities, to determine whether the actual impacts produced by these operations exceed the levels estimated in our analyses above, and ensure that the proposed activities do not result in adverse effects to ESA-listed species. Additionally, we are requiring the reporting of all oil spills (regardless of size) related to the operation of the SPOT DWP and TGL DWP, to ensure that the effects anticipated and analyzed in this Opinion are not exceeded over the life of the projects.

NMFS has determined that the following RPMs are necessary and appropriate to monitor and document any potential impacts or incidental take of ESA-listed species related to the proposed action. In order to ensure that any incidental take of ESA-listed species is detected and monitored, the following reporting measures are required by this ITS:

RPM 1.
The MARAD shall require through the terms of the Deepwater Port License that SPOT and TGL provide annual reports to NMFS, detailing any inadvertent spills or releases of oil from, or in any way related to, the operation of the 2 projects.

RPM 2.
The USCG and MARAD shall require through the terms of the Deepwater Port License that SPOT and TGL conduct comprehensive hydroacoustic monitoring of the pile driving activities at each of their respective offshore terminal construction sites.

9.4 Terms and Conditions

The following terms and conditions are required to implement the above monitoring and reporting measures:

1. To implement RPM No. 1, SPOT and TGL shall each monitor and document any inadvertent spills or releases of oil resulting from, or in any way related to, the operation of their respective projects. SPOT and TGL shall each provide annual reports to NMFS SERO PRD, that include details of any such spills or releases, and any remediation or mitigation measures that were taken in response to those spills or releases. NMFS will track these reports and calculate total spill amounts for each project, on a 5-year basis starting with commencement of operation, to determine if either project is exceeding the levels of oil spill impacts anticipated and analyzed in this Opinion. Reports shall reference the NMFS ECO tracking number for this consultation (SERO-2020-00075), and shall be submitted to the following email address: nmfs.ser.esa.consultations@noaa.gov.

2. To implement RPM No. 2, SPOT and TGL must each develop a comprehensive hydroacoustic monitoring plan shall be developed by each applicant based on the procedures described in Appendix II Procedures for Measuring Pile Driving Sound, which is an appendix to CalTrans' Technical Guidance for the Assessment of Hydroacoustic Effects of Pile Driving on Fish (October 2020). Each applicant must submit its hydroacoustic monitoring plan to NMFS SERO PRD for approval prior to commencement of pile driving activities for the proposed projects.

3. To implement RPM No. 2, SPOT and TGL must submit a detailed report of the final results of such monitoring shall be submitted to NMFS SERO PRD upon completion of the pile driving activities. Reports shall reference the NMFS ECO tracking number for this consultation (SERO-2020-00075), and shall be submitted to the following email address: nmfs.ser.esa.consultations@noaa.gov.

10 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs federal agencies to utilize their authority to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations identified in Biological Opinions can assist action agencies in implementing their responsibilities under Section 7(a)(1). Conservation recommendations are discretionary activities designed to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

During the DWH emergency response, the USCG coordinated with NMFS to obtain recommendations for avoiding and minimizing adverse effects of response activities to ESA-listed species and critical habitats. The agencies formalized these recommendations as a set of 52 Best Management Practices (BMPs) applicable to particular response activities, species, and their habitats. Many of these BMPs were not developed or implemented until well into the DWH response process. The following conservation recommendations are discretionary measures that NMFS believes are consistent with the federal agencies' obligation to avoid or minimize impacts to ESA-listed species in the marine environment from future oil spills and response actions and NMFS strongly recommends that these conservation recommendations be implemented by MARAD and the USCG throughout all future large-scale contaminant spill response actions in the Gulf of Mexico:

1. All response vessel crew members must be instructed to watch for and avoid collisions with wildlife. Report all turtle sightings and all distressed or dead birds, sharks, rays, and marine mammals to the appropriate state hotline.

2. Retrieve all injured/dead/oiled sea turtles using the turtle At-Sea Retrieval Protocol (http://sero.nmfs.noaa.gov/sustainable_fisheries/gulf_sa/turtle_sawfish_release/documents/pdfs/turtle_release_protocols.pdf).

3. If skimming, avoid skimming Sargassum that is not oiled or is only very lightly oiled.

4. If a sea turtle is observed trapped or entangled in boom, open the boom carefully until the animal leaves on its own.

5. Install, monitor, and remove underwater equipment/boom to prevent fish/wildlife entrapment.

6. Do not block major egress points in channels, rivers, passes, and bays.

7. All deployed boom must include: (1) gaps between boom or sufficient space under boom to allow sea turtles to go around or under them, (2) boom should be monitored daily for sea turtle presence.

8. When conducting in situ burning, sea turtle observers must be present on the ignition vessel to spot and retrieve any sea turtles prior to the burn.

9. An additional survey should be conducted in the burn area after the burn is complete.

10. Avoid burning unoiled/lightly oiled *Sargassum*.

11. Arial surveys of all dispersant application areas must be conducted prior to application, and no dispersant application shall occur within 2 nmi of sighted marine mammals/sea turtles.

12. Turtle excluder devices (TEDs) should be installed on all trawl nets.

13. For net recovery of tar balls, a maximum allowable tow time of 30 minutes. After 30 minutes, check the net for any live or dead sea turtles.

14. All vessels must be equipped with the necessary equipment (dip nets, holding containers, towels, etc.) to capture and hold sea turtles aboard the vessel.

15. Resuscitate any live, unresponsive sea turtles according to the official sea turtle resuscitation guidelines (https://www.greateratlantic.fisheries.noaa.gov/protected/stranding/disentanglements/turtle/seaturtlehandlingresuscitationv1.pdf).

16. Safely release uninjured and unoiled sea turtles over the stern of the boat, when gear is not in use, the engine is in neutral, and in areas where they are unlikely to be recaptured or injured by vessels.

17. All vessels shall operate at "no wake/idle" speed at all times while in water where the draft of the vessel provides less than a 4-foot clearance from the bottom. All vessels shall follow deep-water routes whenever possible.

18. Avoid scouring and prop-scarring submerged aquatic vegetation (e.g., seagrass).

19. Natural Resource Advisors (NRAs), or agency biologists should accompany all cleanup crews (both daytime and nighttime operations) in appropriate numbers to ensure BMPs are implemented properly. Contact the section 7 Coordinator/Liaison for recommendations on appropriate numbers.

11 REINITIATION OF CONSULTATION

This concludes formal consultation on the proposed actions. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if: (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action on listed species or designated critical habitat in a manner or to an extent not considered in this Opinion, (3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this Opinion, or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the USCG and MARAD must immediately request reinitiation of formal consultation and project activities may only resume if the USCG and MARAD establish that such continuation will not violate sections 7(a)(2) and 7(d) of the ESA.

12 LITERATURE CITED

35 FR 18319. 1970. List of endangered foreign fish and wildlife. Federal Register 35(233):18319-18322.

66 FR 67495. 2001. Sea Turtle Conservation; Restrictions Applicable to Fishing and Scientific Research Activities. Final Rule. Federal Register 66(250):67495-67496.

69 FR 40734. 2004. Atlantic Highly Migratory Species (HMS); Pelagic Longline Fishery. Final Rule. Federal Register 69(128):40734-40758.

70 FR 42508. 2005. Sea Turtle Conservation; Exceptions to Taking Prohibitions for Endangered Sea Turtles. Federal Register 70(141):42508-42510.

71 FR 45428. 2006. Fisheries of the Caribbean, Gulf of Mexico, and South Atlantic; Reef Fish Fishery of the Gulf of Mexico; Amendment 18A. Final Rule. Federal Register 71(153):45428-45436.

72 FR 43176. 2007. Sea Turtle Conservation; Observer Requirement for Fisheries. Final Rule. Federal Register 72(149):43176-43186.

80 FR 15271. 2015. Endangered and Threatened Species; Identification and Proposed Listing of Eleven Distinct Population Segments of Green Sea Turtles (*Chelonia mydas*) as Endangered or Threatened and Revision of Current Listings. Federal Register 80(55):15272-15337.

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.

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.

Adler-Fenchel, H. S. 1980. Acoustically derived estimate of the size distribution for a sample of sperm whales (*Physeter catodon*) in the western North Atlantic. Canadian Journal of Fisheries and Aquatic Sciences 37(12):2358-2361.

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.

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.

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

Allen, M. R., H. de Coninck, O. P. Dube, and D. J. Heogh-Guldberg Ove; Jacob, Kejun; Revi, Aromar; Rogelj, Joeri; Roy, Joyashree; Shindell, Drew; Solecki, William; Taylor, Michael; Tschakert, Petra; Waisman, Henri; Halim, Sharina Abdul; Antwi-Agyei, Philip; Aragón-Durand, Fernando; Babiker, Mustafa; Bertoldi, Paolo; Bindi, Marco; Brown, Sally; Buckeridge, Marcos; Camilloni, Ines; Cartwright, Anton; Cramer, Wolfgang; Dasgupta, Purnamita; Diedhiou, Arona; Djalante, Riyanti; Dong, Wenjie; Ebi, Kristie L.; Engelbrecht, Francois; Fifita, Solomone; Ford, James; Forster, Piers; Fuss, Sabine; Hayward, Bronwyn; Hourcade, Jean-Charles; Ginzburg, Veronika; Guiot, Joel; Handa, Collins; Hijioka, Yasuaki; Humphreys, Stephen; Kainuma, Mikiko; Kala, Jatin; Kanninen, Markku; Kheshgi, Haroon; Kobayashi, Shigeki; Kriegler, Elmar; Ley, Debora; Liverman, Diana; Mahowald, Natalie; Mechler, Reinhard; Mehrotra, Shagun; Mulugetta, Yacob; Mundaca, Luis; Newman, Peter; Okereke, Chukwumerije; Payne, Antony; Perez, Rosa; Pinho, Patricia Fernanda; Revokatova, Anastasia; Riahi, Keywan; Schultz, Seth; Séférian, Roland; Seneviratne, Sonia I.; Steg, Linda; Suarez Rodriguez, Avelino G.; Sugiyama, Taishi; Thomas, Adelle; Vilariño, Maria Virginia; Wairiu, Morgan; Warren, Rachel; Zhou, Guangsheng; Zickfeld, Kirsten. 2018. Technical Summary. In: *Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* [V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield (eds.)].

Amos, A. F. 1989. The occurrence of Hawksbills (*Eretmochelys imbricata*) along the Texas Coast. Pages 9-11 *in* S. A. Eckert, K. L. Eckert, and T. H. Richardson, editors. Ninth Annual Workshop on Sea Turtle Conservation and Biology.

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.

Au, W. W. L., and M. Green. 2000. Acoustic interaction of humpback whales and whale-watching boats. Marine Environmental Research 49(5):469-481.

Avens, L., J. C. Taylor, L. R. Goshe, T. T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8(3):165-177.

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

Baker, C. S., L. M. Herman, B. G. Bays, and G. B. Bauer. 1983. The impact of vessel traffic on the behavior of humpback whales in southeast Alaska: 1982 season. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, National Marine Mammal Laboratory.

Baker, C. S., A. Perry, and G. Vequist. 1988. Humpback whales of Glacier Bay, Alaska. Whalewatcher 22(3):13-17.

Baker, J., C. Littnan, and D. Johnston. 2006a. 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.

Baker, J. D., C. L. Littnan, and D. W. Johnston. 2006b. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research 2:21-30.

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. 1985a. Impact of ocean debris on marine turtles: Entanglement and ingestion Pages 387-429 *in* R. S. Shomura, and H. O. Yoshida, editors. Workshop on the Fate and Impact of Marine Debris, Honolulu, Hawaii.

Balazs, G. H. 1985b. Impact of ocean debris on marine turtles: Entanglement and ingestion Pages 387-429 *in* R. S. Shomura, and H. O. Yoshida, editors. Workshop on the Fate and Impact of Marine Debris, Honolulu, Hawaii.

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

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

Bauer, G. B. 1986. The behavior of humpback whales in Hawaii and modifications of behavior induced by human interventions. University of Hawaii.

Bauer, G. B., and L. M. Herman. 1986. Effects of vessel traffic on the behavior of humpback whales in Hawaii. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Honolulu, Hawaii.

Baum, J., E. Medina, J. A. Musick, and M. Smale. 2006. *Carcharhinus longimanus*. 2011 IUCN Red List of Threatened Species. International Union for Conservation of Nature and Natural Resources.

Baumgartner, M. F., and B. R. Mate. 2003. Summertime foraging ecology of North Atlantic right whales. Marine Ecology Progress Series 264:123-135.

Baumgartner, M. F., K. D. Mullin, L. N. May, and T. D. Leming. 2001. Cetacean habitats in the northern Gulf of Mexico. Fishery Bulletin 99(2):219-239.

Beale, C. M., and P. Monaghan. 2004. Human disturbance: people as predation-free predators? Journal of Applied Ecology 41:335-343.

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

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

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

Berenshtein, I., and coauthors. 2020. Invisible oil beyond the Deepwater Horizon satellite footprint. Science Advances 6(7):eaaw8863.

Best, P. B. 1979. Social organization in sperm whales, *Physeter macrocephalus*. Pages 227-289 *in* H. E. Winn, and B. L. Olla, editors. Behavior of Marine Animals: Current Perspectives in Research, volume 3 Cetaceans. Plenum Press, New York.

Best, P. B., and D. S. Butterworth. 1980. Timing of oestrus within sperm whale schools. Report of the International Whaling Commission Special Issue 2:137-140.

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. 1997. Foraging ecology and nutrition of sea turtles. Pages 199–231 *in* The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.

Bjorndal, K. A., and A. B. Bolten. 2002. Proceedings of a workshop on assessing abundance and trends for in-water sea turtle populations. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, NMFS-SEFSC-445.

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.

BOEM. 2014. User's Guide for the 2014 Gulfwide Offshore Activities Data System (GOADS-2014). Bureau of Ocean Energy anagement, US DOI, New Orleans, LA.

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.

Bond, J. M. 1999. Genetic analysis of the sperm whale (Physeter macrocephalus) using microsatellites.

Bonfil, R. 2009. The biology and ecology of the silky shark, *Carcharhinus falciformis*. Pages 114-127 *in* T. J. Pitcher, editor. Sharks of the Open Ocean. Blackwell Publishing Ltd., Oxford, UK.

Bonfil, R., and coauthors. 2008. The biology and ecology of the oceanic whitetip shark, *Carcharhinus longimanus*. Sharks of the Open Ocean: Biology, Fisheries, and Conservation:128-139.

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

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.

Boulan, R. H., Jr. 1983. Some notes on the population biology of green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtles in the northern U.S. Virgin Islands: 1981-1983. Report to the National Marine Fisheries Service, Grant No. NA82-GA-A-00044.

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

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

Bowen, B. W., and W. N. Witzell. 1996. Proceedings of the International Symposium on Sea Turtle Conservation Genetics. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, NMFS-SEFSC-396.

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

Brainard, R. E., and coauthors. 2011. Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center, NOAA Technical Memorandum NMFS-PIFSC-27, Honolulu, HI.

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.

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

Bresette, M., 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.

Burks, C., K. D. Mullin, S. L. Swartz, and A. Martinez. 2001. Cruise results: NOAA ship *Gordon Gunter* Cruise GU-OI-01 (11) 6 February - 3 April 2001. Marine mammal survey of Puerto Rico and The Virgin Islands, and a study of sperm whales in the southeastern Gulf of Mexico. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

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.

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

Carr, A. 1987. Impact of nondegradable marine debris on the ecology and survival outlook of sea turtles. Marine Pollution Bulletin 18(6B):352-356.

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.

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

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

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 10(11):e02936.

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

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

Chaloupka, M., 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 C. J. Limpus. 1997. Robust statistical modelling of hawksbill sea turtle growth rates (southern Great Barrier Reef). Marine Ecology Progress Series 146(1-3):1-8.

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

Chassot, E., M. Amandè, C. Pierre, R. Pianet, and R. Dédo. 2008. Some preliminary results on tuna discards and bycatch in the French purse seine fishery of the eastern Atlantic Ocean. Collective Volume Of Scientific Papers 64.

Cheng, L., and coauthors. 2017. Improved estimates of ocean heat content from 1960 to 2015. Science Advances 3(3):e1601545.

Chin, A., P. Kyne, T. Walker, and R. McAuley. 2010. An integrated risk assessment for climate change: Analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. Global Change Biology 16:1936-1953.

Chiquet, R. A., B. Ma, A. S. Ackleh, N. Pal, and N. Sidorovskaia. 2013. Demographic analysis of sperm whales using matrix population models. Ecological Modelling 248:71-79.

Christal, J., H. Whitehead, and E. Lettevall. 1998. Sperm whale social units: Variation and change. Canadian Journal of Zoology 76(8):1431-1440.

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.

Clapham, P. J., and D. K. Mattila. 1993. Reactions of humpback whales to skin biopsy sampling on a West Indies breeding ground. Marine Mammal Science 9(4):382-391.

Clapham, P. J., S. B. Young, and R. L. Brownell Jr. 1999. Baleen whales: Conservation issues and the status of the most endangered populations. Mammal Review 29(1):35-60.

Clark, C., and coauthors. 2009a. Acoustic masking of baleen whale communications: Potential impacts from anthropogenic sources. Pages 56 *in* Eighteenth Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada.

Clark, C. W., and coauthors. 2009b. Acoustic masking in marine ecosystems as a function of anthropogenic sound sources. International Whaling Commission Scientific Committee, Madeira, Portugal.

Clark, C. W., and coauthors. 2009c. Acoustic masking in marine ecosystems: Intuitions, analysis, and implication. Marine Ecology Progress Series 395:201-222.

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.

Clarke, M. R. 1962. Stomach contents of a sperm whale caught off Madeira in 1959. Norsk Hvalfangst-Tidende 51(5):173-189, 191.

Clarke, M. R. 1976. Observations on sperm whale diving. Journal of the Marine Biological Association of the United Kingdom 56(3):809-810.

Clarke, M. R. 1979. The head of the sperm whale. Scientific American 240(1):128-132, 134, 136-141.

Coelho, R., and coauthors. 2009. Notes of the reproduction of the oceanic whitetip shark, Carcharhinus longimanus, in the southwestern equatorial Atlantic Ocean. . Pages 1734-1740 *in*.

Compagno, L. J. V. 1984. Part 2. Carcharhiniformes. Pages 251-655 *in* FAO Species Catalogue. Sharks of the World. An Annotated and Illustrated Catalogue of Sharks Species Known to Date, volume 4. FAO.

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

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

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

Cortés, E. 1999. Standardized diet compositions and trophic levels of sharks. ICES Journal of Marine Science: Journal du Conseil 56(5):707-717.

Cortés, E., and coauthors. 2010. Ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. . Aquatic Living Resources 23:25-34.

Cortés, E., and coauthors. 2012. Expanded ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. .

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.

Crabbe, M. J. 2008. Climate change, global warming and coral reefs: modelling the effects of temperature. Computational Biology and Chemistry 32(5):311-4.

Cranford, T. W. 1992. Functional morphology of the odontocete forehead: Implications for sound generation. University of California, Santa Cruz.

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

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.

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

Davis, R. W., W. E. Evans, and B. Wursig. 2000. Cetaceans, sea turtles, and seabirds in the northern Gulf of Mexico: Distribution, abundance, and habitat associations. Volume II: Technical Report. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Davis, R. W., and G. S. Fargion. 1996. Distribution and abundance of cetaceans in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. U.S. Department of the Interior, Minerals Management Service.

Davis, R. W., and coauthors. 1998. Physical habitat of cetaceans along the continental slope of the north-central and western Gulf of Mexico. Marine Mammal Science 14(3):490-507.

Davis, R. W., and coauthors. 2002. Cetacean habitat in the northern oceanic Gulf of Mexico. Deep Sea Research Part I: Oceanographic Research Papers 49(1):121-142.

Deakos, M. H. 2010. Ecology and social behavior of a resident manta ray (*Manta alfredi*) population of Maui, Hawai'i. Dissertation. Univeristy of Hawai'i at Mānoa, Honolulu, HI.

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.

Derraik, J. G. B. 2002. The pollution of the marine environment by plastic debris: A review. Marine Pollution Bulletin 44:842-852.

Dewar, H., and coauthors. 2008. Movements and site fidelity of the giant manta ray, *Manta birostris*, in the Komodo Marine Park, Indonesia. Marine Biology 155(2):121-133.

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

Diez, C. E., and R. P. Van Dam. 2007. In-water surveys for marine turtles at foraging grounds of Culebra Archipelago, Puerto Rico

Dillon, M. C., H. Whitehead, and J. M. Wright. 1997. Geographical population structure of female sperm whales assessed by mitochondrial DNA variation. European Research on Cetaceans 10:302.

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).

Doney, S. C., and coauthors. 2012. Climate change impacts on marine ecosystems. Marine Science 4.

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.

Dufault, S., and H. Whitehead. 1995. The geographic stock structure of female and immature sperm whales in the South Pacific. Report of the International Whaling Commission 45:401-405.

Dufault, S., H. Whitehead, and M. Dillon. 1999. An examination of the current knowledge on the stock structure of sperm whales (*Physeter macrocephalus*) worldwide. Journal of Cetacean Research and Management 1:1-10.

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.

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

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

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/.

Dwyer, K. L., C. E. Ryder, and R. Prescott. 2003. Anthropogenic mortality of leatherback turtles in Massachusetts waters. Pages 260 *in* J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation, Miami, Florida.

Eckert, K. L. 1995. Hawksbill sea turtle (*Eretmochelys imbricata*). Pages 76-108 *in* National Marine Fisheries Service and U.S. Fish and Wildlife Service Status Reviews for Sea Turtles Listed under the Endangered Species Act of 1973. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Springs, Maryland.

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

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

Eckert, K. L., J. A. Overing, and B. B. Lettsome. 1992. Sea turtle recovery action plan for the British Virgin Islands. UNEP Caribbean Environment Programme, Wider Caribbean Sea Turtle Recovery Team and Conservation Network, Kingston, Jamaica.

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

Eckert, S. A. 1989. Diving and foraging behavior of the leatherback sea turtle, *Dermochelys coriacea*. University of Georgia, Athens, Georgia.

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

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

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

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

Eguchi, T., P. H. Dutton, S. A. Garner, and J. Alexander-Garner. 2006. Estimating juvenile survival rates and age at first nesting of leatherback turtles at St. Croix, U.S. Virgin Islands. Pages 292-293 *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.

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.

Engelhaupt, D., and coauthors. 2009. Female philopatry in coastal basins and male dispersion across the North Atlantic in a highly mobile marine species, the sperm whale (*Physeter macrocephalus*). Molecular Ecology 18(20):4193-4205.

Engelhaupt, D. T. 2004. Phylogeography, kinship and molecular ecology of sperm whales (*Physeter macrocephalus*). University of Durham.

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.

Evans, P. G. H., and A. Bjørge. 2013. Impacts of climate change on marine mammals. Marine Climate Change Impacts Parternship: Science Review:134-148.

FAO. 2012. Fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-Exploited Aquatic Species. FAO Fisheries and Aquaculture Report No. 1032, Rome.

Farmer, N. A., and coauthors. 2022. Modeling protected species distributions and habitats to inform siting and management of pioneering ocean industries: A case study for Gulf of Mexico aquaculture. bioRxiv:2022.04.07.487536.

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

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.

Fleming, E. H. 2001. Swimming Against the Tide: Recent Surveys of Exploitation, Trade, And Management of Marine Turtles In the Northern Caribbean. TRAFFIC North America, Washington, D.C., USA.

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 Mexico Science 25(2):131-143.

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

Frasier, K. 2020. Evaluating Impacts of Deep Oil Spills on Oceanic Marine Mammals. Pages 419-441 *in*.

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.

Frédou, F. L., and coauthors. 2015. Sharks caught by the Brazilian tuna longline fleet: an overview. Rev. Fish Biol. Fish. 25:365-377.

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

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

Gaither, M. R., B. W. Bowen, L. A. Rocha, and J. C. Briggs. 2016. Fishes that rule the world: circumtropical distributions revisited. Fish and Fisheries 17(3):664-679.

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

Garcia M., D., and L. Sarti. 2000. Reproductive cycles of leatherback turtles. Pages 163 *in* F. A. Abreu-Grobois, R. Briseno-Duenas, R. Marquez, and L. Sarti, editors. Eighteenth International Sea Turtle Symposium.

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

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.

Gauthier, J., and R. Sears. 1999. Behavioral response of four species of balaenopterid whales to biopsy sampling. Marine Mammal Science 15(1):85-101.

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., 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).

Gero, S., and H. Whitehead. 2007. Suckling behavior in sperm whale calves: Observations and hypotheses. Marine Mammal Science 23(2):398-413.

GESAMP. 1990. The State of the Marine Environment. Reports and Studies, GESAMP, London.

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

Girondot, M., and coauthors. 2015. Spatio-temporal distribution of *Manta birostris* in French Guiana waters. Journal of the Marine Biological Association of the United Kingdom 95(1):153-160.

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.

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

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

Goold, J. C., and S. E. Jones. 1995. Time and frequency domain characteristics of sperm whale clicks. Journal of the Acoustical Society of America 98(3):1279-1291.

Gordon, J., R. Leaper, F. G. Hartley, and O. Chappell. 1992. Effects of whale-watching vessels on the surface and underwater acoustic behaviour of sperm whales off Kaikoura, New Zealand. Department of Conservation, Science & Research Series No. 52, Wellington, New Zealand.

Gordon, J. C. D. 1987. The behaviour and ecology of sperm whales off Sri Lanka. University of Cambridge, Cambridge.

Gower, J. F. R., and S. A. King. 2011. Distribution of floatingSargassumin the Gulf of Mexico and the Atlantic Ocean mapped using MERIS. International Journal of Remote Sensing 32(7):1917-1929.

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).

Graham, T. R. 2009. Scyphozoan jellies as prey for leatherback sea turtles off central California. Master's Theses. San Jose State University.

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.

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

Gregory, M. R. 2009. Environmental implications of plastic debris in marine settings-entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Philosophical Transactions of the Royal Society of London B Biological Sciences 364(1526):2013-2025.

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

Groombridge, B., and R. Luxmoore. 1989. The Green Turtle and Hawksbill (Reptilia: Cheloniidae): World Status, Exploitation and Trade. Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, Lausanne, Switzerland.

Gudger, E. W. 1922. The most northerly record of the capture in Atlantic waters of the United States of the giant ray, *Manta birostris*. Science 55(1422):338-340.

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.

Hall, J. D. 1982. Prince William Sound, Alaska: Humpback whale population and vessel traffic study. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, Juneau Management Office, Contract No. 81-ABG-00265., Juneau, Alaska.

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

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

Hatch, L. T., C. W. Clark, S. M. V. Parijs, A. S. Frankel, and D. W. Ponirakis. 2012. Quantifying loss of acoustic communication space for right whales in and around a US. National Marine Sanctuary. Conservation Biology 26(6):983-994.

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.

Hayhoe, K., and coauthors. 2018. In *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II* (Reidmiller, D.R., et al. [eds.]). U.S. Global Change Research Program, Washington, DC, USA.

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

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

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

Hazen, E. L., and coauthors. 2012. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change 3(3):234-238.

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. 2003a. 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 T. R. Menzel. 1999. Life table analysis of long-lived marine species with implications for conservation and management. Pages 137-148 *in* American Fisheries Society Symposium.

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

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.

Heron, S. F., C. M. Eakin, J. A. Maynard, and R. van Hooidonk. 2016. Impacts and effects of ocean warming on coral reefs. Pages 177-197 *in* D. Laffoley, and J. M. Baxter, editors. Explaining Ocean Warming: Causes, scale, effects and consequences. IUCN, Gland, Switzerland.

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 Mexico. Pages 447-453 *in* K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D. C.

Hildebrand, J. A. 2009. Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology Progress Series 395:5-20.

Hillis, Z.-M., and A. L. Mackay. 1989. Research report on nesting and tagging of hawksbill sea turtles *Eretmocheys imbricata* at Buck Island Reef National Monument, U.S. Virgin Islands, 1987-88.

Hilterman, M., E. Goverse, M. Godfrey, M. Girondot, and C. Sakimin. 2003. Seasonal sand temperature profiles of four major leatherback nesting beaches in the Guyana Shield. Pages 189-190 *in* J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation.

Hinojosa-Alvarez, S., R. P. Walter, P. Diaz-Jaimes, F. Galván-Magaña, and E. M. Paig-Tran. 2016. A potential third Manta Ray species near the Yucatán Peninsula? Evidence for a recently diverged and novel genetic Manta group from the Gulf of Mexico. PeerJ 4:e2586.

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

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.

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

Hooker, S. K., R. W. Baird, S. Al-Omari, S. Gowans, and H. Whitehead. 2001. Behavioral reactions of northern bottlenose whales (Hyperoodon ampullatus) to biopsy darting and tag attachment procedures. Fishery Bulletin 99(2):303-308.

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

Howey-Jordan, L. A., and coauthors. 2013. Complex Movements, Philopatry and Expanded Depth Range of a Severely Threatened Pelagic Shark, the Oceanic Whitetip (Carcharhinus longimanus) in the Western North Atlantic. PLoS ONE 8(2):e56588.

Howey, L. A., and coauthors. 2016. Into the deep: the functionality of mesopelagic excursions by an oceanic apex predator. Ecology and Evolution 6(15):5290-5304.

Hu, C., and coauthors. 2016. Sargassum coverage in the northeastern Gulf of Mexico during 2010 from Landsat and airborne observations: Implications for the Deepwater Horizon oil spill impact assessment. Marine Pollution Bulletin 107(1):15-21.

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

Intergovernmental Panel on Climate Change. 2013. Climate Change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change, Cambridge, United Kingdom; New York, NY.

IOTC. 2014. Report of the Seventeenth Session of the IOTC Scientific Committee. .

IPCC. 2014. Climate change 2014: Impacts, adaptation, and vulnerability. IPCC Working Group II contribution to AR5. Intergovernmental Panel on Climate Change.

Isojunno, S., and P. J. O. Miller. 2015. Sperm whale response to tag boat presence: biologically informed hidden state models quantify lost feeding opportunities. Ecosphere 6(1).

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.

Jahoda, M., and coauthors. 2003. Mediterranean fin whale's (*Balaenoptera physalus*) response to small vessels and biopsy sampling assessed through passive tracking and timing of respiration. Marine Mammal Science 19(1):96-110.

Jambeck, J. R., and coauthors. 2015. Plastic waste inputs from land into the ocean. Science 347(6223):768-771.

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

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

Jaquet, N. 2006. A simple photogrammetric technique to measure sperm whales at sea. Marine Mammal Science 22(4):862-879.

Jaquet, N., S. Dawson, and E. Slooten. 1998. Diving behavior of male sperm whales: Foraging implications. International Whaling Commission Scientific Committee, Muscat.

Jay, A., and coauthors. 2018. In: *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II* [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA:33-71.

Jefferson, T. A., and A. J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Review 27(1):27-50.

Jochens, A., and coauthors. 2008. Sperm whale seismic study in the Gulf of Mexico: Synthesis report. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, OCS Study MMS 2008-006, New Orleans, Louisiana.

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.

Johnston, M. A., and coauthors. 2017. Long-Term Monitoring at East and West Flower Garden Banks: 2016 Annual Report. . Pages 132 *in* Marine Sanctuaries Conservation Series U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary, Galveston, TX.

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.

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

Joung, S. J., N. F. Chen, H. H. Hsu, and K. M. Liu. 2016. Estimates of life history parameters of the oceanic whitetip shark,*Carcharhinus longimanus*, in the Western North Pacific Ocean. . Mar. Biol. Res.:1-11.

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.

Kasuya, T. 1991. Density dependent growth in North Pacific sperm whales. Marine Mammal Science 7(3):230-257.

Katsanevakis, S. 2008. Marine debris, a growing problem: Sources distribution, composition, and impacts. Pages 53-100 *in* T. N. Hofer, editor. Marine Pollution: New Research. Nova Science Publishers, Inc, New York.

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

Kintisch, E. 2006. As the seas warm: Researchers have a long way to go before they can pinpoint climate-change effects on oceangoing species. Science 313:776-779.

Koehler, N. 2006. Humpback whale habitat use patterns and interactions with vessels at Point Adolphus, southeastern Alaska. University of Alaska, Fairbanks, Fairbanks, Alaska.

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.

Laist, D. W. 1987. Overview of the biological effects of lost and discarded plastic debris in the marine environment. Marine Pollution Bulletin 18(6):319-326.

Laist, D. W. 1997. Impacts of marine debris: Entanglement of marine life in marine debris including a comprehensive list of species with entanglement and ingestion records. Pages 99-140 *in* J. M. Coe, and D. B. Rogers, editors. Marine Debris: Sources, Impacts, and Solutions. Springer-Verlag, New York, New York.

Lambertsen, R. H., B. A. Kohn, J. P. Sundberg, and C. D. Buergelt. 1987. Genital papillomatosis in sperm whale bulls. Journal of Wildlife Diseases 23(3):361-367.

Lapointe, B. E. 1986. Phosphorus-limited photosynthesis and growth of Sargassum natans and Sargassum fluitans (Phaeophyceae) in the western North Atlantic. Deep Sea Research Part A. Oceanographic Research Papers 33(3):391-399.

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

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

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.

Learmonth, J. A., and coauthors. 2006. Potential effects of climate change on marine mammals. Oceanography and Marine Biology: an Annual Review 44:431-464.

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

León, Y. M., and C. E. Diez. 2000. Ecology and population biology of hawksbill turtles at a Caribbean feeding ground. Pages 32-33 *in* F. A. Abreu-Grobois, R. Briseño-Dueñas, R. Márquez-Millán, and L. Sarti-Martinez, editors. Eighteenth International Sea Turtle Symposium. U.S. Department of Commerce, Mazatlán, Sinaloa, México.

Lesage, V., C. Barrette, M. C. S. Kingsley, and B. Sjare. 1999. The Effect of Vessel Noise on the Vocal Behavior of Belugas in the St. Lawrence River Estuary, Canada. Marine Mammal Science 15(1):65-84.

Lessa, R., F. M. Santana, and R. Paglerani. 1999. Age, growth and stock structure of the oceanic whitetip shark, *Carcharhinus longimanus*, from the southwestern equatorial Atlantic. . Fisheries Research 42:21-30.

Levenson, C. 1974. Source level and bistatic target strength of the sperm whale (*Physeter catodon*) measured from an oceanographic aircraft. Journal of the Acoustical Society of America 55(5):1100-1103.

Lezama, C. 2009. impacto de la pesqueria artesanal sobre la tortoga verde (*Chelonia mydas*) en las costas del Rio de la Plata exterior. Universidad de la República.

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

Limpus, C. J. 1992. The hawksbill turtle, *Eretmochelys imbricata*, in Queensland: Population struture within a southern Great Barrier Reef feeding ground. Australian Wildlife Research 19:489-506.

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

Lockyer, C. 1981. Estimates of growth and energy budget for the sperm whale, *Physeter catodon*. Pages 489-504 *in* J. Gordon Clark, editor. Mammals in the Seas volume 3:

General papers and large cetaceans. Food and Agriculture Organization of the United Nations, Rome.

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

López-Mendilaharsu, M., A. Estrades, M. A. C. Caraccio, V., M. Hernández, and V. Quirici. 2006. Biología, ecología yetología de las tortugas marinas en la zona costera uru-guaya, Montevideo, Uruguay: Vida Silvestre, Uruguay.

Lopez-Pujol, J., and M.-X. Ren. 2009. Biodiversity and the Three Gorges Reservoir: A troubled marriage. Journal of Natural History 43(43-44):2765-2786.

Love, M. S., A. Baldera, C. Yeaung, and C. Robbins. 2013. The Gulf of Mexico Ecosystem: A coastal & marine atlas. Ocean Conservancy, Gulf Restoration Center, New Orleans, LA.

Lund, F. P. 1985. Hawksbill turtle (*Eretmochelys imbricata*) nesting on the East Coast of Florida. Journal of Herpetology 19(1):166-168.

Lusseau, D. 2004. The hidden cost of tourism: Detecting long-term effects of tourism using behavioral information. Ecology and Society 9(1):2.

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.

Lutz, P. L., Musick, J.A., & Wyneken, J. 2003. The Biology of Sea Turtles, Volume II (1st ed.). CRC Press.

Lyrholm, T., and U. Gyllensten. 1998. Global matrilineal population structure in sperm whales as indicated by mitochondrial DNA sequences. Proceedings of the Royal Society of London Series B Biological Sciences 265(1406):1679-1684.

Lyrholm, T., O. Leimar, B. Johanneson, and U. Gyllensten. 1999. Sex-biased dispersal in sperm whales: Contrasting mitochondrial and nuclear genetic structure of global populations. Transactions of the Royal Society of London, Series B: Biological Sciences 266(1417):347-354.

Mackay, A. L. 2006. 2005 sea turtle monitoring program the East End beaches (Jack's, Isaac's, and East End Bay) St. Croix, U.S. Virgin Islands. Nature Conservancy.

MacLeod, C. D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. Endangered Species Research 7(2):125-136.

MacLeod, C. D., and coauthors. 2005. Climate change and the cetacean community of north-west Scotland. Biological Conservation 124(4):477-483.

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

Malme, C. I., P. R. Miles, C. W. Clark, P. Tyack, and J. E. Bird. 1983. Investigations of the potential effects of underwater noise from petroleum industry activities on migrating gray whale behavior. Final report for the period of 7 June 1982 - 31 July 1983. Department of the Interior, Minerals Management Service, Alaska OCS Office, Anchorage, Alaska.

MantaMatcher. 2016. Manta Matcher - The Wildbook for Manta Rays.

Marcovaldi, N., B. B. Gifforni, H. Becker, F. N. Fiedler, and G. Sales. 2009. Sea Turtle Interactions in Coastal Net Fisheries in Brazil. U.S. National Marine Fisheries Service, Southeast Fisheries Science Center: Honolulu, Gland, Switze, Honolulu, Hawaii, USA.

Markowitz, T., Richter, C., Gordon, J., Sagnol, O., Markowitz, W., Macaulay, J., Isojunno, S., & Fernandes, M. 2011. Effects of tourism on the behaviour of sperm whales inhabiting the Kaikōura Canyon. Kaikōura Sperm Whales and Tourism Research Project.

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., and coauthors. 2011. *Manta birostris*. The IUCN Red List of Threatened Species.

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.

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

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

Mayor, P. A., B. Phillips, and Z.-M. Hillis-Starr. 1998. Results of the stomach content analysis on the juvenile hawksbill turtles of Buck Island Reef National Monument, U.S.V.I. Pages 230-233 *in* S. P. Epperly, and J. Braun, editors. Seventeenth Annual Sea Turtle Symposium.

Maze-Foley, K., and K. D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. Journal of Cetacean Research and Management 8(2):203-213.

McCauley, S., and K. Bjorndal. 1999. Conservation implications of dietary dilution from debris ingestion: Sublethal effects in post-hatchling loggerhead sea turtles. Conservation Biology 13(4):925-929.

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

McDonald, T. L., and S. P. Powers. 2015. Estimates of Sargassum Extent in Four Regions of the Northern Gulf of Mexico from Aerial Surveys. DWH NRDA Water Column Technical Working Group Report.

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

McMahon, C. R., and G. C. Hays. 2006. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology 12(7):1330-1338.

McMichael, E., R. R. Carthy, and J. A. Seminoff. 2003. Evidence of homing behavior in juvenile green turtles in the northeastern Gulf of Mexico. 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.

Mesnick, S., M. Anderson, C. Chan, A. Allen, and A. Dixson. 2005. Phylogenetic analysis of testes size in cetaceans: Using primate models to test predictions of sperm competition theory. Pages 191 *in* Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, California.

Mesnick, S. L., and coauthors. 1999. Culture and genetic evolution in whales. Science 284(5423):2055-2059.

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

Meylan, A., and M. Donnelly. 1999. Status justification for listing the hawksbill turtle (*Eretmochelys imbricata*) as critically endangered on the 1996 IUCN Red List of threatened animals. Chelonian Conservation and Biology 3(2):200-224.

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

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

Meylan, A. B., 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.

Miller, J. D. 1997. Reproduction in sea turtles. Pages 51-58 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.

Miller, M. H., and C. Klimovich. 2017. Endangered Species Act status review report: Giant manta ray (*Manta birostris*) and reef manta ray (*Manta alfredi*). U.S. Department of Commerce, National Oceanic and Atmoshperic Administration, National Marine Fisheries Servcie, Office of Protected Resources, Silver Spring, MD.

Milliken, T., and H. Tokunaga. 1987. The Japanese sea turtle trade 1970-1986. TRAFFIC (JAPAN), Center for Environmental Education, Washington, D. C.

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

Mo, C. L. 1988. Effect of bacterial and fungal infection on hatching success of Olive Ridley sea turtle eggs. World Wildlife Fund-U.S.

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

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

Moncada Gavilan, F. 2001. Status and distribution of the loggerhead turtle, *Caretta caretta*, in the wider Caribbean region. Pages 36-40 *in* K. L. Eckert, and F. A. Abreu Grobois, editors. Marine Turtle Conservation in the Wider Caribbean Region - A Dialogue for Effective Regional Management, Santo Domingo, Dominican Republic.

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.

Mortimer, J. A., and coauthors. 2003. Growth rates of immature hawksbills (*Eretmochelys imbricata*) at Aldabra Atoll, Seychelles (Western Indian Ocean). Pages 247-248 *in* J. A. Seminoff, editor Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation.

Mortimer, J. A., M. Day, and D. Broderick. 2002. Sea turtle populations of the Chagos Archipelago, British Indian Ocean Territory. Pages 47-49 *in* A. Mosier, A. Foley, and B. Brost, editors. Twentieth Annual Symposium on Sea Turtle Biology and Conservation.

Mortimer, J. A., and M. Donnelly. 2008. Hawksbill turtle (*Eretmochelys imbricata*) International Union for Conservation of Nature and Natural Resources.

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.

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

Mullin, K., and coauthors. 1991. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. Pages 48 *in* Ninth Biennial Conference on the Biology of Marine Mammals, Chicago, Illinois.

Mullin, K. D., and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico, 1996-2001. Marine Mammal Science 20(4):787-807.

Mullin, K. D., and coauthors. 1994. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. Fishery Bulletin 92(4):773-786.

Mullins, J., H. Whitehead, and L. S. Weilgart. 1988. Behaviour and vocalizations of two single sperm whales, *Physeter macrocephalus*, off Nova Scotia. Canadian Journal of Fisheries and Aquatic Sciences 45(10):1736-1743.

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.

Musyl, M. K., and coauthors. 2011. Postrelease survival, vertical and horizontal movements, and thermal habitats of five species of pelagic sharks in the central Pacific Ocean. . Fishery Bulletin 109:341- 368.

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

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

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. Estimated impacts of mortality reductions on loggerhead sea turtle population dynamics, preliminary results. Presented at the meeting of the Reef Fish Management Committee of the Gulf of Mexico Fishery Management Council. Gulf of Mexico Fishery Management Council, Tamps, FL.

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. 2005. Endangered Species Act Section 7 Consultation - Biological Opinion on the continued authorization of reef fish fishing under the Gulf of Mexico Reef Fish Fishery Management Plan and Proposed Amendment 23.

NMFS. 2010. Recovery plan for the sperm whale *(Physeter macrocephalus)*. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, Maryland.

NMFS. 2011. Endangered Species Act Section 7 Consultation - Biological Opinion on the Continued Authorization of Reef Fish Fishing under the Gulf of Mexico (Gulf) Reef Fish Fishery Management Plan (RFFMP). Sumbitted on September 30, 2011, St. Petersburg, Florida.

NMFS. 2014. Reinitiation of Endangered Species Act (ESA) Section 7 Consultation on the Continued Implementation of the Sea Turtle Conservation Regulations under the ESA and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act. NOAA. NMFS, Southeast Regional Office, Protected Resources Division.

NMFS. 2015. Sperm Whale (Physeter macrocephalus) 5-Year Review: Summary and Evaluation June 2015. National Marine Fisheries Service, Office of Protected Resources

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, 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 Mexico. 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 Mexico. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, St. Petersburg, Florida.

NMFS, and USFWS. 1995. Status reviews for sea turtles listed under the Endangered Species Act of 1973. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS, and USFWS. 1998a. Recovery plan for U. S. Pacific populations of the hawksbill turtle (*Eretmochelys imbricata*). National Marine Fisheires Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 1998b. Recovery plan for U.S. Pacific populations of the leatherback turtle (*Dermochelys coriacea*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS, and USFWS. 2007a. 5-year review: Summary and evaluation, green sea turtle (*Chelonia mydas*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service.

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

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

NMFS, and USFWS. 2007d. 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. 2007e. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2007f. 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. 2008a. 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, 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.

NMFS, and USFWS. 2013a. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2013b. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2015. 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, USFWS, and SEMARNAT. 2011a. Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. Pages 156 *in*. National Marine Fisheries Service, Silver Spring, Maryland.

NMFS, USFWS, and SEMARNAT. 2011b. 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, Washington, D.C.

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

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

NMFS and USFWS. 2008c. 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. 2013. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. NOAA, National Marine Fisheries Service, Office of Protected Resources and U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Office.

Norris, K. S., and G. W. Harvey. 1972. A theory for the function of the spermaceti organ of the sperm whale (*Physeter catodon* L.). Pages 397-417 *in* S. R. Galler, K. Schmidt-Koenig, G. J. Jacobs, and R. E. Belleville, editors. Animal Orientation and Navigation. National Air and Space Administration, Washington, D. C.

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.

O'Malley, M. P., K. Lee-Brooks, and H. B. Medd. 2013. The global economic impact of manta ray watching tourism. PLOS ONE 8(5):e65051.

O'Malley, M. p., K. A. Townsend, P. Hilton, S. Heinrichs, and J. D. Stewart. 2017. Characterization of the trade in manta and devil ray gill plates in China and South-east Asia through trader surveys. Aquatic Conservation: Marine and Freshwater Ecosystems 27(2):394-413.

Odell, D. K. 1992. Sperm whale, *Physeter macrocephalus*. Pages 168-175 *in* S. R. Humphrey, editor. Rare and Endangered Biota of Florida, volume Volume 1: Mammals. University Press of Florida, Gainesville, Florida.

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.

Oliver, S., M. Braccini, S. J. Newman, and E. S. Harvey. 2015. Global patterns in the bycatch of sharks and rays. Marine Policy 54:86-97.

Palacios, D. M., and B. R. Mate. 1996. Attack by false killer whales (*Pseudorca crassidens*) on sperm whales (*Physeter macrocephalus*) in the Galápagos Islands. Marine Mammal Science 12(4):582-587.

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

Papastavrou, V., S. C. Smith, and H. Whitehead. 1989. Diving behaviour of the sperm whale, *Physeter macrocephalus*, off the Galapagos Islands. Canadian Journal of Zoology 67(4):839-846.

Parks, S. E., C. W. Clark, and P. L. Tyack. 2007. Short- and long-term changes in right whale calling behavior: The potential effects of noise on acoustic communication. Journal of the Acoustical Society of America 122(6):3725-3731.

Parks, S. E., M. Johnson, D. Nowacek, and P. L. Tyack. 2011. Individual right whales call louder in increased environmental noise. Biology Letters 7(1):33-35.

Parks, S. E., M. P. Johnson, D. P. Nowacek, and P. L. Tyack. 2012. Changes in vocal behavior of North Atlantic right whales in increased noise. Pages 4 *in* A. N. Popper, and A. Hawkings, editors. The Effects of Noise on Aquatic Life. Springer Science.

Parsons, E. C. M. 2012. The Negative Impacts of Whale-Watching. Journal of Marine Biology 2012:1-9.

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

Payne, P. M., J. R. Nicolas, L. O'brien, and K. D. Powers. 1986. The distribution of the humpback whale, Megaptera novaeangliae, on Georges Bank and in the Gulf of Maine in relation to densities of the sand eel, Ammodytes americanus. Fishery Bulletin 84(2):271-277.

Payne, P. M., and coauthors. 1990. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in prey abundance. Fishery Bulletin 88(4):687-696.

Pecl, G. T., and G. D. Jackson. 2008. The potential impacts of climate change on inshore squid: Biology, ecology and fisheries. Reviews in Fish Biology and Fisheries 18:373-385.

Perry, S. L., D. P. Demaster, and G. K. Silber. 1999. The sperm whales (*Physeter macrocephalus*). Marine Fisheries Review 61(1):59-74.

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.

Plotkin, P., and A. F. Amos. 1990. Effects of anthropogenic debris on sea turtles in the northwestern Gulf of Mexico. Pages 736-743 *in* R. S. Shoumura, and M. L. Godfrey, editors. Proceedings of the Second Internntional Conference on Marine Debris. NOAA Technical Memorandum NMFS SWFSC-154. U.S. Department of Commerce, Honolulu, Hawaii.

Plotkin, P. T. 2003. Adult migrations and habitat use. Pages 225-241 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, volume 2. CRC Press.

Plotkin, P. T., and A. F. Amos. 1988. Entanglement in and ingestion of marine debris by sea turtles stranded along the South Texas coast. Pages 7 *in* Supplemental Deliverables under Entanglement-Debris Task No. 3. Debris, Entanglement and Possible Causes of Death in Stranded Sea Turtles (FY88).

Polyakov, I. V., V. A. Alexeev, U. S. Bhatt, E. I. Polyakova, and X. Zhang. 2009. North Atlantic warming: patterns of long-term trend and multidecadal variability. Climate Dynamics 34(3-Feb):439-457.

Powers, S. P., F. J. Hernandez, R. H. Condon, J. M. Drymon, and C. M. Free. 2013. Novel pathways for injury from offshore oil spills: Direct, sublethal and indirect effects of the *Deepwater Horizon* oil spill on pelagic *Sargassum* communities. PLoS ONE 8(9):e74802.

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

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

Pritchard, P. C. H., and P. Trebbau. 1984. The turtles of Venezuela. SSAR.

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

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 Mexico. University of Miami Press, Coral Gables, Florida.

Reeves, R. R., and H. Whitehead. 1997. Status of the sperm whale, *Physeter macrocephalus*, in Canada. Canadian Field-Naturalist 111(2):15.

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

Rice, D. W. 1989. Sperm whale *Physeter macrocephalus* Linnaeus, 1758. Pages 177-234 *in* S. H. Ridgway, and R. Harrison, editors. Handbook of Marine Mammals, volume 4: River Dolphins and the Larger Toothed Whales. Academic Press, San Diego, California.

Rice, J., and S. Harley. 2012. Stock assessment of oceanic whitetip sharks in the western and central Pacific Ocean. Western and Central Pacific Fisheries Commission Scientific Committee Eighth Regular Session.WCPFC-SC8-2012/SA-WP-06 Rev 1., 53. Pages 53 *in*.

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

Richardson, W. J. 1995. Marine mammal hearing. Pages 205-240 *in* W. J. Richardson, C. R. J. Greene, C. I. Malme, and D. H. Thomson, editors. Marine Mammals and Noise. Academic Press, San Diego, California.

Richardson, W. J., C. R. Greene, and B. Wursig, editors. 1985. Behavior, disturbance responses and distribution of bowhead whales (*Balaena mysticetus*) in the eastern Beaufort Sea, 1980-84: A summary. LGL Ecological Research Associates, Inc., Bryan, Texas.

Richter, C., S. Dawson, and E. Slooten. 2006. Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 22(1):46-63.

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

Rivas-Zinno, F. 2012. Captura incidental de tortugas marinas en Bajos del Solis, Uruguay. Universidad de la Republica Uruguay, Departamento de Ecologia y Evolucion.

Roberts, J. J., and coauthors. 2016. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. Scientific Reports 6:22615.

Robinson, R. A., and coauthors. 2005. Climate change and migratory species. Defra Research, British Trust for Ornithology, Norfolk, U.K. .

Roden, C. L., and K. D. Mullin. 2000. Sightings of cetaceans in the northern Caribbean Sea and adjacent waters, winter 1995. Caribbean Journal of Science 36(3-4):280-288.

Rodrigues, J., D. Freitas, Í. Fernandes, and R. Lessa. 2015. Estrutura populacional do tubarao estrangeiro (*Carcharhinus longimanus*) no Atlantico Sul. 3.

Rolland, R. M., and coauthors. 2017. Fecal glucocorticoids and anthropogenic injury and mortality in North Atlantic right whales Eubalaena glacialis. Endangered Species Research 34:417-429.

Rolland, R. M., and coauthors. 2012. Evidence that ship noise increases stress in right whales. Proceedings of the Royal Society of London Series B Biological Sciences 279(1737):2363-2368.

Rolland, R. M., and coauthors. 2016. Health of North Atlantic right whales Eubalaena glacialis over three decades: from individual health to demographic and population health trends. Marine Ecology Progress Series 542:265-282.

Romanov, E. V. 2002. Bycatch in the tuna purse-seine fisheries of the western Indian Ocean. Fishery Bulletin 100(1):90-105.

Rosel, P. E., and coauthors. 2016. Status review of Bryde's whales (Balaenoptera edeni) in the Gulf of Mexico under the Endangered Species Act.

Rosel, P. E., L. A. Wilcox, T. K. Yamada, and K. D. Mullin. 2021. A new species of baleen whale (Balaenoptera) from the Gulf of Mexico, with a review of its geographic distribution. Marine Mammal Science 37(2):577-610.

Rubin, R. D., K. R. Kumli, and G. Chilcott. 2008. Dive characteristics and movement patterns of acoustic and satellite-tagged manta rays (*Manta birostris*) in the Revillagigedos Islands of Mexico. American Elasmobranch Society, Montreal, Canada.

Sakai, H., H. Ichihashi, H. Suganuma, and R. Tatsukawa. 1995. Heavy metal monitoring in sea turtles using eggs. Marine Pollution Bulletin 30(5):347-353.

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

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

Scheidat, M., A. Gilles, K.-H. Kock, and U. Siebert. 2006. Harbour porpoise (Phocoena phocoena) abundance in German waters (July 2004 and May 2005). International Whaling Commission Scientific Committee, St. Kitts and Nevis, West Indies.

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.

Schulz, J. P. 1975. Sea turtles nesting in Surinam. Zoologische Verhandelingen 143:3-172.

Seki, T., T. Taniuchi, H. Nakano, and M. Shimizu. 1998. Age, growth and reproduction of the oceanic whitetip Shark from the Pacific Ocean. . Fisheries Science 64:14-20.

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

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

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

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

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

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

Simmonds, M. P., and W. J. Eliott. 2009. Climate change and cetaceans: Concerns and recent developments. Journal of the Marine Biological Association of the United Kingdom 89(1):203-210.

Simmonds, M. P., and S. J. Isaac. 2007. The impacts of climate change on marine mammals: Early signs of significant problems. Oryx 41(1):19-26.

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

Soldevilla, M. S., A. J. Debich, L. P. Garrison, J. A. Hildebrand, and S. M. Wiggins. 2022. Rice's whales in the northwestern Gulf of Mexico: call variation and occurrence beyond the known core habitat. Endangered Species Research 48:155-174.

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

Sparks, T. D., J. C. Norris, R. Benson, and W. E. Evans. 1995. Distribution of sperm whales in the northwestern Gulf of Mexico as determined from an acoustic survey. Pages 108 *in* Eleventh Biennial Conference on the Biology of Marine Mammals, Orlando, Florida.

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

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

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

Stapleton, S., and C. Stapleton. 2006. Tagging and nesting research on hawksbill turtles (*Eretmochelys imbricata*) at Jumby Bay, Long Island, Antigua, West Indies: 2005 annual report. Jumby Bay Island Company, Ltd.

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

Starbird, C. H., and M. M. Suarez. 1994. Leatherback sea turtle nesting on the north Vogelkop coast of Irian Jaya and the discovery of a leatherback sea turtle fishery on Kei Kecil Island. Pages 143-146 *in* K. A. Bjorndal, A. B. Bolten, D. A. Johnson, and P. J. Eliazar, editors. Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.

Stewart, J. D., and coauthors. 2016a. Spatial ecology and conservation of *Manta birostris* in the Indo-Pacific. Biological Conservation 200:178-183.

Stewart, J. D., E. M. Hoyos-Padilla, K. R. Kumli, and R. D. Rubin. 2016b. Deep-water feeding and behavioral plasticity in *Manta birostris* revealed by archival tags and submersible observations. Zoology 119.

Stewart, J. D., M. Nuttall, E. L. Hickerson, and M. A. Johnston. 2018. Important juvenile manta ray habitat at Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. Marine Biology 165:111.

Stewart, K., and C. Johnson. 2006. *Dermochelys coriacea*—Leatherback sea turtle. Chelonian Research Monographs 3:144-157.

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

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

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

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

Strasburg, D. W. 1958. Distribution, abundance, and habits of pelagic sharks in the central Pacific Ocean. Fisheries 1:2S.

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

Tambourgi, M., and coauthors. 2013. Reproductive aspects of the oceanic whitetip shark, Carcharhinus longimanus (Elasmobranchii: Carcharhinidae), in the equatorial and southwestern Atlantic Ocean. . Brazilian Journal of Oceanography 61:161-168.

Tennessen, J. B., and S. E. Parks. 2016. Acoustic propagation modeling indicates vocal compensation in noise improves communication range for North Atlantic right whales. Endangered Species Research 30:225-237.

TEWG. 1998. 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. 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. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. 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.

Tiwari, M., B. P. Wallace, and M. Girondot. 2013. *Dermochelys coriacea* (Northwest Atlantic Ocean subpopulation). The IUCN Red List of Threatened Species (e.T46967827A46967830. http://dx.doi.org/10.2305/IUCN.UK.2013-2.RLTS.T46967827A46967830.en).

Tolotti, M. T., P. Bach, F. Hazin, P. Travassos, and L. Dagorn. 2015. Vulnerability of the Oceanic Whitetip Shark to Pelagic Longline Fisheries. PLoS ONE PLoS ONE 10(10).

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

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

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

Trustees, D. H. N. 2016. Deepwater Horizon Oil Spill: Final Programmatic Damage Assessment and Restoration Plan (PDARP) and Final Programmatic Environmental Impact Statement. NOAA, http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan.

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

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.

U.S. Coast Guard, M. A., Environmental Resource Management. 2022. Final Environmental Impact Statement for SPOT Terminal LLC Deepwater Port License Application. D. o. H. Security, editor, Washington D.C.

van Dam, R., and L. Sarti. 1989. Sea turtle biology and conservation on Mona Island, Puerto Rico. Report for 1989.

Van Dam, R., L. Sarti M., and D. Pares J. 1991. The hawksbills of Mona Island, Puerto Rico: Report for 1990. Sociedad Chelonia and Departmento. Recursos Naturales, Puerto Rico.

Van Dam, R. P., and C. E. Diez. 1997. Predation by hawksbill turtles on sponges at Mona Island, Puerto Rico. Pages 1421-1426 *in* Eighth International Coral Reef Symposium.

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

Venables, S. 2013. Short term behavioural responses of manta rays, *Manta alfredi*, to tourism interactions in Coral Bay, Western Australia. Thesis. Murdoch University.

Walker, W. A., and J. M. Coe. 1990. Survey of marine debris ingestion by odontocete cetaceans. Pages 747-774 *in* Second International Conference on Marine Debris, Honolulu, Hawaii.

Wallace, B., and coauthors. 2015. Estimating degree of oiling of sea turtles and surface habitat during the Deepwater Horizon oil spill: implications for injury quantification.(ST_TR. 02). DWH Sea Turtles NRDA Technical Working Group Report.

Waring, G. T., Elizabeth Josephson, Katherine Maze-Foley, Patricia E. Rosel. 2016. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments-2015. NMFS Northeast Fisheries Science Center, NFMS-NE-238, Woods Hole, Massachusetts.

Waring, G. T., and coauthors. 1997. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments -- 1996. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., and coauthors. 2002. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments -- 2002. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service.

Watkins, W. A. 1980. Acoustics and the behavior of sperm whales. Pages 283-290 *in* R.-G. Busnel, and J. F. Fish, editors. Animal Sonar Systems. Plenum Press, New York and London.

Watkins, W. A., K. Moore, and P. Tyack. 1985a. Codas shared by Caribbean sperm whales. Pages 81 *in* Sixth Biennial Conference on the Biology of Marine Mammals, Vancouver, B.C., Canada.

Watkins, W. A., K. E. Moore, and P. L. Tyack. 1985b. Sperm whale acoustic behaviors in the southeast Caribbean. Cetology 49:1-15.

Watkins, W. A., K. E. Moore, D. Wartzok, and J. H. Johnson. 1981. Radio tracking of finback (*Balaenoptera physalus*), and humpback (*Megaptera novaeangliae*) whales in Prince William Sound, Alaska, USA. Deep Sea Research Part I: Oceanographic Research Papers 28(6):577-588.

Watkins, W. A., and W. E. Schevill. 1975. Sperm whales (*Physeter catodon*) react to pingers. Deep Sea Research 22:123-129.

Watkins, W. A., and W. E. Schevill. 1977. Sperm whale codas. Journal of the Acoustical Society of America 62(6):1485-1490.

Weilgart, L. S. 2007. The impacts of anthropogenic ocean noise on cetaceans and implications for management. Canadian Journal of Zoology 85:1091-1116.

Weilgart, L. S., and H. Whitehead. 1988. Distinctive vocalizations from mature male sperm whales (*Physeter macrocephalus*). Canadian Journal of Zoology 66(9):1931-1937.

Weilgart, L. S., and H. Whitehead. 1993. Coda communication by sperm whales (*Physeter macrocephalus*) off the Galapagos Islands. Canadian Journal of Zoology 71(4):744-752.

Weilgart, L. S., and H. Whitehead. 1997. Group-specific dialects and geographical variation in coda repertoire in South Pacific sperm whales. Behavioral Ecology and Sociobiology 40(5):277-285.

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.

Weller, D., and coauthors. 1996. Observations of an interaction between sperm whales and short-finned pilot whales in the Gulf of Mexico. Marine Mammal Science 12(4):588-594.

Weller, D. W., B. Wursig, S. K. Lynn, and A. J. Schiro. 2000. Preliminary findings on the occurrence and site fidelity of photo-identified sperm whales (*Physeter macrocephalus*) in the northern Gulf of Mexico. Gulf of Mexico Science 18(1):35-39.

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.

Whitehead, H. 1996. Babysitting, dive synchrony, and indications of alloparental care in sperm whales. Behavioral Ecology and Sociobiology 38(4):237-244.

Whitehead, H. 2002. Estimates of the current global population size and historical trajectory for sperm whales. Marine Ecology Progress Series 242:295-304.

Whitehead, H. 2003. Sperm Whales: Social Evolution in the Ocean. University of Chicago Press.

Whitehead, H., J. Christal, and S. Dufault. 1997. Past and distant whaling and the rapid decline of sperm whales off the Galapagos Islands. Conservation Biology 11(6):1387-1396.

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

Wiggins, S. M., J. M. Hall, B. J. Thayre, and J. A. Hildebrand. 2016. Gulf of Mexico low-frequency ocean soundscape impacted by airguns. The Journal of the Acoustical Society of America 140(1):176-183.

Wilkinson, C. 2004. Status of Coral Reefs of the World: 2004. Australian Institute of Marine Science, ISSN 1447-6185.

Williams, R., and coauthors. 2011. Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. Conservation Letters 4(3):228-233.

Williamson, M. J., A. S. Kavanagh, M. J. Noad, E. Kniest, and R. A. Dunlop. 2016. The effect of close approaches for tagging activities by small research vessels on the behavior of humpback whales (Megaptera novaeangliae). Marine Mammal Science.

Willis-Norton, E., and coauthors. 2015. Climate change impacts on leatherback turtle pelagic habitat in the Southeast Pacific. Deep Sea Research Part II: Topical Studies in Oceanography 113:260-267.

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. .

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

Witt, M. J., B. J. Godley, A. C. Broderick, R. Penrose, and C. S. Martin. 2006. Leatherback turtles, jellyfish and climate change in the northwest Atlantic: Current situation and possible future scenarios. Pages 356-357 *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.

Witzell, W. N. 1983. Synopsis of biological data on the hawksbill sea turtle, *Eretmochelys imbricata* (Linnaeus, 1766). Food and Agricultural Organization of the United Nations, Rome.

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

Wright, A., and coauthors. 2007. Anthropogenic noise as a stressor in animals: A multidisciplinary perspective. International Journal of Comparative Psychology.

Wursig, B., T. A. Jefferson, and D. J. Schmidly. 2000. The Marine Mammals of the Gulf of Mexico. Texas A&M University Press, College Station, Texas.

Wursig, B., S. K. Lynn, T. A. Jefferson, and K. D. Mullin. 1998. Behaviour of cetaceans in the northen Gulf of Mexico relative to survey ships and aircraft. Aquatic Mammals 24(1):41-50.

Young, C. N., and coauthors. 2016a. Status review report: oceanic whitetip shark (Carcharhinius longimanus). . DOC National Oceanic and Atmospheric Administration.

Young, C. N., and coauthors. 2016b. Status review report: oceanic whitetip shark (*Carcharhinius longimanus*). Final Report to the National Marine Fisheries Service, Office of Protected Resources. .

Young, C. N., and coauthors. 2017. Status review report: oceanic whitetip shark (*Carcharhinius longimanus*). Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Silver Spring, Maryland.

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

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

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.