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Endangered Species Act Section 7
Biological Opinion

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List of Acronyms

AFWTA – Atlantic Fleet Weapons Training Area

AGRRA – Atlantic and Gulf Rapid Reef Assessment

ANSI – American National Standards Institute

ARMS - Autonomous Reef Monitoring Structures

ATON – Aids-to-Navigation

BA – Biological Assessment

BIP – Blow-in-Place

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act

CFMC – Caribbean Fishery Management Council

CITES – Convention on International Trade in Endangered Species of Wild Fauna and Flora

CRCP – Coral Reef Conservation Program

DGM – Digital Geophysical Mapping

DO – Dissolved Oxygen

DOI – Department of Interior

DPS – Distinct Population Segment

DWH – Deepwater Horizon

ECA – Eastern Conservation Area

EMA – Eastern Maneuver Area

EPA – U.S. Environmental Protection Agency

ESA – Endangered Species Act

ESTCP - Environmental Security Technology Certification Program

EZ – Exclusion Zone

FFWCC – Florida Fish and Wildlife Conservation Commission

FMP – Fishery Management Plan

GMI – GeoMarine, Inc.

GPS – Global Positioning System

ITS – Incidental Take Statement

LIA – Live Impact Area

MC – Munitions Constituents

MEC – Munitions and Explosives of Concern

MMPA – Marine Mammal Protection Act

MPA – Marine Protected Area

MPPEH – Material Potentially Presenting an Explosive Hazard

NASD – Naval Ammunition Support Detachment

NCCOS – National Centers for Coastal Ocean Science

NCRMP – National Coral Reef Monitoring Program

NIWC - Naval Information Warfare Center

NMFS – National Marine Fisheries Service

NOAA – National Oceanic and Atmospheric Administration

NPL – National Priority List

NRC – National Research Council

NTCRA – Non-Time-Critical Removal Action

OB/OD – Open Burn/Open Detonation

OPR – Office of Protected Resources

PBF – Physical and Biological Features

PDC – Project Design Criteria

POCIS – Polar Organic Chemical Integrative Samplers

PRD – Protected Resources Division

PRDNER – Puerto Rico Department of Natural and Environmental Resources

PREQB – Puerto Rico Environmental Quality Board

PTS – Permanent Threshold Shift

ROV – Remotely Operated Vehicle

RPM – Reasonable and Prudent Measure

SIA – Surface Impact Area

SE – Standard Error

SEL – Sound Exposure Level

SERO – Southeast Regional Office

SOP – Standard Operating Procedure

SPL – Sound Pressure Level

SUXOS/DS – Senior Unexploded Ordnance Supervisor/Diving Supervisor

SWMU – Solid Waste Management Unit

TEWG – Technical Expert Working Group

TL – Total Length

TTS – Temporary Threshold Shift

USCG – U.S. Coast Guard

USFWS – U.S. Fish and Wildlife Service

USVI – U.S. Virgin Islands

UXO – Unexploded Ordnance

VNTR – Vieques Naval Training Range

VNWR – Vieques National Wildlife Refuge

WAA – Wide Area Assessment

1 INTRODUCTION

The Endangered Species Act of 1973 (ESA), as amended (16 U.S.C. 1531 et seq.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat they depend on. Section 7(a)(2) of the ESA requires Federal agencies to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Federal agencies must do so in consultation with National Marine Fisheries Service (NMFS) for threatened or endangered species (ESA-listed), or designated critical habitat that may be affected by the action that are under NMFS jurisdiction (50 C.F.R. §402.14(a)). If a Federal action agency determines that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS concur with that determination for species under NMFS jurisdiction, consultation concludes informally (50 C.F.R. §402.14(b)).

Section 7(b)(3) of the ESA requires that at the conclusion of consultation NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat. If NMFS determines that the action is likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If an incidental take is expected, section 7(b)(4) requires NMFS to provide an incidental take statement (ITS), which exempts take incidental to an otherwise lawful action, and specifies the impact of any incidental taking, including reasonable and prudent measures (RPMs) to minimize such impacts and terms and conditions to implement the RPMs.

Updates to the regulations governing interagency consultation (50 C.F.R. 402) became effective on October 28, 2019 (84 FR 44976). This consultation was pending at the time the regulations became effective and we are applying the updated regulations to the consultation. As the preamble to the final rule adopting the regulations noted, "This final rule does not lower or raise the bar on section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and consistency, streamlines consultations, and codifies existing practice." We have reviewed the information and analyses relied upon to complete this biological opinion (Opinion) in light of the updated regulations and conclude the Opinion is fully consistent with the updated regulations.

The Federal action agency for this consultation is the U.S. Department of the Navy (Navy). The Navy proposes the investigation and the implementation of removal/remedial actions to address underwater munitions offshore of the former Naval Ammunition Support Detachment (NASD) and the former Vieques Naval Training Range (VNTR) in Vieques, Puerto Rico, collectively identified as underwater ordnance (UXO) 16. This programmatic consultation consults on activities by the Navy that will be conducted in phases over an approximately 20-year period throughout UXO 16.

Programmatic Consultations

NMFS and the U.S. Fish and Wildlife Service (USFWS) have developed a range of techniques to streamline the procedures and time involved in consultations for broad agency programs or numerous similar activities with predictable effects on listed species and critical habitat. Some of the more common of these techniques and the requirements for ensuring that streamlined consultation procedures comply with section 7 of the ESA and its implementing regulations are discussed in the October 2002 joint Services memorandum Alternative Approaches for Streamlining Section 7 Consultation on Hazardous Fuels Treatment Projects (see also 68 FR 1628 [January 13, 2003] for the notice of availability of the memorandum).

A programmatic consultation is a consultation addressing an agency's multiple actions on a program, region, or other basis usually over an extended period of time. Programmatic consultations allow the Services to consult on the effects of programmatic actions such as: (1) multiple similar, frequently occurring or routine actions expected to be implemented in particular geographic areas; and (2) a proposed program, plan, policy, or regulation providing a framework for future proposed actions (84 FR 44976, August 27, 2019). A programmatic consultation should identify project design criteria (PDCs) or standards that will be applicable to all future projects implemented under the program. PDCs serve to prevent adverse effects to listed species, or to limit adverse effects to predictable levels that will not jeopardize the continued existence of listed species or destroy or adversely modify critical habitat. Avoidance and minimization of adverse effects to species and their designated critical habitat is accomplished by implementing PDCs at the individual project level or taken together from all projects under the programmatic consultation. For those activities that meet the PDCs, there is no need for project-specific consultations. For actions that do not meet the PDCs or for which specifics of individual activities are not yet known, step-down consultations are needed under a programmatic consultation. The following elements should be included in a programmatic consultation to ensure its consistency with ESA section 7 and its implementing regulations:

1. Description of the manner in which activities to be implemented under the programmatic consultation may affect listed species and critical habitat and evaluation of expected level of adverse effects from covered projects;

2. PDCs to prevent or limit future adverse effects on listed species and designated critical habitat;

3. Process for evaluating and tracking expected and actual aggregate (net) additive effects of all projects expected to be addressed under the programmatic consultation. The programmatic consultation document must demonstrate that when the PDCs are applied to each project, the aggregate effect of all projects would not jeopardize listed species or destroy or adversely modify critical habitat;

4. Procedures for streamlined step-down consultation. As discussed above, if an approved programmatic consultation document is sufficiently detailed, step-down consultations ideally will consist of certifications, concurrences, or a streamlined opinion between action agency biologists and consulting agency biologists. An action agency biologist or team will provide a description of a proposed project and a certification that it will be implemented in accordance with the PDCs. The action agency also provides a description of anticipated project-specific effects and a tallying of net effects to date resulting from projects implemented under the program, and certification that these effects are consistent with those anticipated in the programmatic consultation. The consulting agency biologist reviews the submission and provides concurrence or an opinion, or offers adjustments to the project necessary to bring it into compliance with the programmatic consultation. The project-specific consultation process must also identify any effects that were not considered in the programmatic consultation and an ITS will be prepared to exempt additional incidental take, if needed, for the step-down, formal consultation. Finally, project-specific consultation procedures must provide contingencies for proposed projects that cannot be implemented in accordance with the PDCs; full stand-alone consultation may be performed on these projects if they are too dissimilar in nature or in expected effects from those projected in the programmatic opinion;

5. Procedures for monitoring projects and validating effects predictions; and

6. Comprehensive review of the program, generally conducted annually.

A framework programmatic action is a federal action that approves a framework for the development of future actions that are authorized, funded, or carried out later. In a step-down tiered approach under the framework programmatic action, which is what will be used here, the programmatic consultation establishes an analytical and standardized framework so that future step-down consultations, if necessary, may occur at the implementation or authorization stage when the effects are better known and thus the consultation will be more effective and efficient. The Services promulgated changes to the section 7(a)(2) implementing regulations (80 FR 26832, May 11, 2015) (ITS rule) that define two types of programmatic actions addressing certain types of policies, plans, regulations, and programs. In this type of programmatic action, any take of ESA-listed species would not occur unless and until those future actions are authorized, funded, or carried out and subject to a separate step-down consultation, as appropriate. At that time, an ITS may be issued, if necessary, to exempt incidental take causes by those specific actions. The second type of programmatic action, known as a mixed programmatic action, such as the Navy's phased investigation and removal/remedial activities within UXO 16, combines direct approval of actions that will not be subject to further ESA section 7(a)(2) consultation and approval of a framework for the development of future actions that are authorized, funded, or carried out at a later time. For mixed programmatic actions, as defined in the 2015 ITS rule at 50 C.F.R. 402.02, NMFS is required to issue an ITS for those portions of the program that are authorized at the program level, not subject to a future section 7 consultation, are reasonably certain to result in incidental take, and are otherwise compliant with ESA section 7(a)(2). In this type of mixed programmatic action, any future actions within the framework that will be subject to step-down consultations when the future actions are authorized, funded, or carried out, an ITS may be issued at that time for the incidental take associated with those actions, as necessary.

This consultation, Opinion, and associated ITS were completed in accordance with ESA section 7, associated implementing regulations (50 C.F.R. §§402.01-402.16), and agency policy and guidance. This consultation was conducted by the NMFS Office of Protected Resources (OPR) Endangered Species Act Interagency Cooperation Division (hereafter referred to as "we" or "our").

This document represents the NMFS opinion on the effects of these actions on giant manta ray (*Manta birostris*); Nassau grouper (*Epinephelus striatus*); oceanic whitetip (*Carcharhinus longimanus*) and scalloped hammerhead sharks (*Sphyrna lewini*; Northwest and Western Central Atlantic Distinct Population Segment [DPS]); lobed star (*Orbicella annularis*), mountainous star (*Orbicella faveolata*), boulder star (*Orbicella franksi*), elkhorn (*Acropora palmata*), staghorn (*Acropora cervicornis*), pillar (*Dendrogyra cylindrus*), and rough cactus corals (*Mycetophyllia ferox*); green (*Chelonia mydas*; North Atlantic and South Atlantic DPSs), leatherback (*Dermochelys coriacea*), hawksbill (*Eretmochelys imbricata*), and loggerhead sea turtles (*Caretta caretta*; Northwest Atlantic Ocean DPS); blue (*Balaenoptera musculus*), fin (*Balaenoptera physalus*), sei (*Balaenoptera borealis*), and sperm whales (*Physeter microcephalus*); and elkhorn and staghorn coral critical habitat.

A complete record of this consultation is on file at the NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

The Navy purchased portions of Vieques Island in the early 1940s in order to conduct activities related to military training. The former Atlantic Fleet Weapons Training Area (AFWTA) was divided into the NASD and the VNTR (Figure 1). In the former NASD on the western end of Vieques, site operations consisted mainly of ammunition loading and storage, vehicle and facility maintenance, and open burn/open detonation (OB/OD). In the former VNTR on the eastern end of Vieques, various naval gunfire training activities were conducted, including air-to-ground ordnance delivery and amphibious landings, and the main base of operations, Camp Garcia, was located here.

In 2001, in accordance with Public Law 106-398, the former NASD was apportioned and transferred to the Department of the Interior (DOI), the Municipality of Vieques, and the Puerto Rico Conservation Trust. The former NASD consists of approximately 8,100 acres (ac). Solid Waste Management Unit (SWMU) 4, which was used as an OB/OD area for the thermal destruction and open detonation of retrograde and surplus munitions, fuels, and propellants, is located in the western portion of the former NASD (Figure 1). The DOI property is managed by the USFWS as part of the Vieques National Wildlife Refuge (VNWR). In 2003, in accordance with Public Law 107-107, the former VNTR was transferred to DOI to be operated by the USFWS as part of the VNWR. The former VNTR consists of approximately 14,600 ac divided into four operational areas comprising, from west to east, the 11,000 ac Eastern Maneuver Area (EMA), the 2,500 ac Surface Impact Area (SIA), the 900 ac Live Impact Area (LIA), and the 200 ac Eastern Conservation Area (ECA; Figure 1). The EMA was established in 1947 to provide military maneuvering areas and ranges for training in amphibious landings, small arms fire, artillery and tank fire, shore fire control, and combat engineering tasks (Figure 1). The ranges in the EMA were used for the following: small arms (Ranges 1 and 2), rifle grenades (Range 3), rockets (Range 4) and grenades (Range 5). The SIA was established in the 1950s when several marine artillery targets were constructed and a bullseye target was constructed for inert bombing in 1969. The LIA was established in 1965 with several targets maintained for aerial bombing, including tanks and vehicles, a simulated railroad tunnel, a simulated ammunition dump, a simulated fuel farm, a simulated airstrip, two simulated surface-to-air missile sites, and a strafing target; several point and aerial targets for ships to practice naval gunfire support; one bullseye target for inert bombing; and an OB/OD area for treatment of retrograde ordnance and open burning of propellants and pyrotechnics. The ECA was established as a conservation area but is adjacent to the LIA and may contain munitions due to skips and/or misses.

UXO 16 comprises approximately 11,500 ac in waters surrounding the former NASD and VNTR. The current UXO 16 site boundary was defined in a letter from PREQB to the Enviornmental Protection Agency (May 26, 2004), and published in the Federal Register on August 13, 2004, and in the Vieques Site Management Plan (CH2M Hill 2018). UXO 16 includes three offshore anchorage areas, Mosquito Pier, the area offshore of SWMU 4, explosives safety arcs and artillery safety fans adjacent to the former VNTR, other offshore areas surrounding the former VNTR, and Cayo la Chiva (Figure 1). The offshore anchorage areas were used as temporary anchorage sites by Navy ships containing munitions used during the training activities at the former AFWTA. Mosquito Pier was used for loading and unloading ordnance from Navy ships. The area offshore of SWMU 4 represents the explosives safety arc of the OB/OD operations area in SWMU 4. There were a series of explosives safety arcs and artillery safety fans associated with gun ranges, gun emplacements, and OB/OD areas for the former VNTR. Other offshore areas include the southern portion of Vieques from Puerto Ferro to the eastern portion of Playa la Chiva (Blue Beach) and the eastern tip of the ECA, which all fall outside historic safety arcs and fans. Cayo la Chiva is a 12-ac island south of the EMA that contained a simulated defense position during a 1950 operation and may have been used for live fire training based on the finding and removal of 5-inch (in) rockets on the cay and in the water around the cay in 2017.

The Navy subdivided the former operational areas on the former NASD and VNTR, as well as the offshore portion, into smaller parcels referred to as UXO sites based on historic use, geographic features and land use for the purposes of prioritization, munitions removal, site characterization, and decision making.

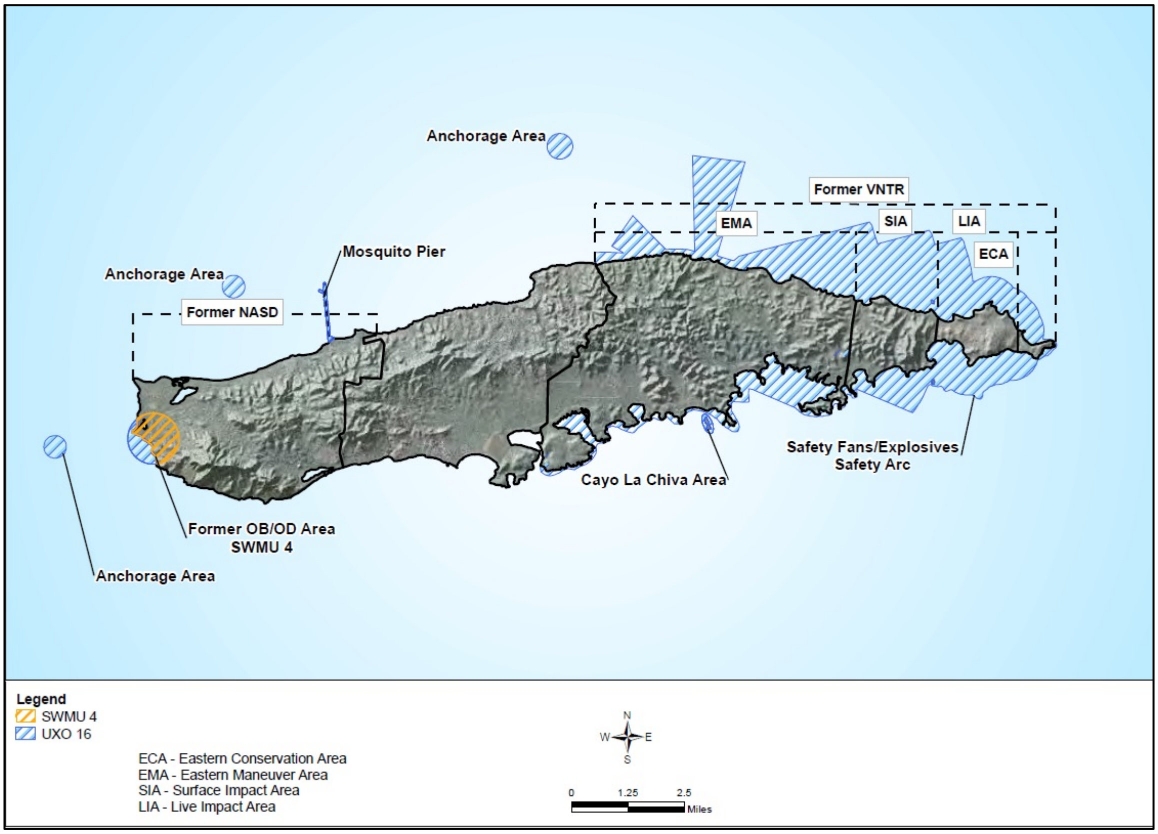


Figure 1. Map of Vieques Island showing the former NASD, VNTR, and water areas in UXO 16 (from CH2M Hill 2018)

The Navy completed a Wide Area Assessment (WAA) using underwater digital geophysical mapping (DGM) to identify locations with concentrations of metallic objects that could indicate the presence of munitions in UXO 16. A supplemental objective of the WAA was to collect video documentation of habitat types and the presence of ESA-listed resources such as corals. A total of 334 survey kilometers (km; 208 miles) were mapped during WAA field activities, providing an effective coverage of 412 acres or 3.6 percent of the 46.5 square kilometers (km^2; 11,500 acres) that comprise UXO 16.

1.2 Consultation History

NMFS Southeast Regional Office (SERO) Protected Resources Division (PRD) began working with the Navy in 2006, providing technical assistance and conducting ESA section 7 consultations for land use controls (including buoy installations), investigations, and removal/remedial activities being conducted by the Navy under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in Vieques. A SERO PRD biologist participated in site inspections to SWMU 4, Bahia Icacos, and Cayo la Chiva between 2006 and 2014, and met with the Navy about conducting a programmatic consultation for underwater activities with the last meeting held January 10, 2017. SERO PRD transferred responsibility for coordination with the Navy, including technical assistance and ESA section 7 consultations related to Navy activities in Vieques to OPR in January 2017.

This Opinion is based on information provided by the Navy, including the *Final: UXO 16 Programmatic Biological Assessment and Essential Fish Habitat Assessment, Atlantic Fleet Weapons Training Area – Vieques Former Naval Ammunition Support Detachment and Former Vieques Naval Training Range, Vieques, Puerto Rico* (CH2MHill 2018). Our communication with the Navy regarding this consultation is summarized as follows:

- **May 31, 2017:** The Navy sent NMFS a letter via email requesting initiation of a programmatic ESA section 7 consultation for underwater activities in UXO 16. The letter was accompanied by an outline describing the proposed content of the Biological Assessment (BA) the Navy was preparing for the consultation. The letter also noted that the Navy planned to submit 7(a)(2), 7(d) analyses and determinations for continued investigation and removal activities in cases where there could be munitions or explosives of concern (MEC) that pose a threat to human health and safety and where operations can be conducted in a way that will not result in take of ESA-listed species or the destruction or adverse modification of designated critical habitat.
- **June 1, 2017:** The Navy sent NMFS a letter via email with their 7(a)(2), 7(d) analysis and determination for a Non-Time-Critical Removal Action (NTCRA) for nine MEC items offshore of Cayo la Chiva.
- **June 15, 2017:** NMFS sent a letter to the Navy responding to their request for initiation of consultation noting that we agreed with their determination regarding the NTCRA for Cayo la Chiva but that we were not initiating consultation because the BA for the proposed activities had not yet been completed and submitted to us.
- **June 22, 2017:** The Navy sent a letter to NMFS via email with their 7(a)(2), 7(d) analysis and determination for the implementation of several underwater activities, including the removal of buoys offshore of Cayo la Chiva due to completion of the NTCRA, barrier retrieval and buoy installation at barrier anchor locations in Bahia Icacos (for which NMFS completed a formal consultation on August 20, 2012, Ref. No. SER-2011-05676), and the retrieval of wave monitoring equipment offshore of several beaches in UXO 16.
- **August 15, 2017:** The Navy, its consultants from CH2MHill, and NMFS participated in a conference call to discuss the programmatic BA, request for initiation of consultation and NMFS response to the request, and the underwater activities included in the Navy's June 22, 2017 letter. The Navy sent NMFS an email the same day with notes from the call and NMFS responded with one correction to the notes via email.
- **February 22, 2018:** The Navy sent a letter via email to NMFS with their 7(a)(2), 7(d) analysis and determination for a NTCRA of encrusted munitions in UXO PI-9 East and adjacent UXO 16. NMFS responded via email the next day indicating that we had

participated in a site inspection of the area where the NTCRA was proposed and had commented on the work plan in the past.

- **May 24, 2018:** The Navy, CH2MHill (now Jacobs), and NMFS met to discuss the draft BA.
- **June 11, 2018:** NMFS provided written comments on the BA to the Navy via email.
- **November 15, 2018:** The Navy, CH2MHill, and NMFS had a conference call to discuss remaining comments on the draft BA.
- **December 21, 2018:** The Navy sent the final programmatic BA requesting initiation of formal consultation. NMFS acknowledged receipt of the BA on the same day and initiated consultation.
- **January 28, 2019:** Consultation was resumed on this day after being held in abeyance for 38 days due to a lapse in appropriations that resulted in a partial government shutdown.
- **March 19, 2019:** The Navy sent a letter via email to NMFS with their 7(a)(2), 7(d) analysis and determination for the implementation of reconnaissance diving to identify underwater munition items for potential future removal actions. The Navy also requested that a draft of the programmatic biological opinion be shared with them once it is ready for review.
- **June 11, 2019:** NMFS sent an email to the Navy with some questions regarding coral sampling, monitoring traps, and sea turtle nesting.
- **June 25, 2019:** The Navy sent a response to NMFS questions via email.
- **July 1, 2019:** The Navy sent a map of areas where USFWS has documented sea turtle nesting around Vieques via email.
- **September 13 and 16, 2019:** The Navy sent Vieques sea turtle nesting information from USFWS to NMFS via email.
- **January 21, 2020:** NMFS sent the Navy the draft programmatic biological opinion for review and comment.
- **February 14, 2020**: The Navy and staff from the Naval Information Warfare Center Pacific (NIWC Pacific) had a conference call with NMFS to introduce and request comments on the proposed Environmental Security Technology Certification Program (ESTCP) funded Coral Ark demonstration project in Vieques. The Navy will add the project to the description of the proposed action for the consultation.
- **April 17, 2020**: The Navy sent NMFS their comments and edits to the draft programmatic biological opinion via email.

2 THE ASSESSMENT FRAMEWORK

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

“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 recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species” (50 C.F.R. §402.02).

“Destruction or adverse modification” means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of an ESA-listed species as a whole (50 C.F.R. §402.02).

This ESA section 7 consultation involves the following steps:

Description of the Action (Section 3): In this programmatic consultation, a description of the action on the part of the Navy, includes those activities that will not require further consultation and those activities for which step-down consultations will be required in the future, if they may affect listed species or designated critical habitat, because the specifics are not known at this time. This section also includes the PDCs for avoidance and minimization of impacts to ESA-listed species and designated critical habitat, and information regarding the procedures for submitting step-down consultation requests and conducting regular reviews under the programmatic consultation.

Action Area (Section 4): We describe the action and those aspects (or stressors) of the action that may have effects on the physical, chemical, and biotic environment. We describe the action area with the spatial extent of the stressors from those actions. Thus, we evaluate the effects vessel transit routes may have on ESA-listed species and designated critical habitat and so include the approximate footprints of these in this consultation as part of the action area.

Stressors Associated with the Action (Section 5): We discuss the potential stressors we expect to result from the action for both the activities that will not require further consultation and for activities that will require step-down consultations.

Status of Species and Designated Critical Habitat (Section 6): We identify the ESA-listed species and designated critical habitat that are likely to co-occur with the stressors from the action in space and time and evaluate the status of those species and habitat. We also identify those *Species and Designated Critical Habitat Not Likely to be Adversely Affected*, detail our effects analysis for these species and critical habitats (Section 7.1), and identify the status of the *Species and Designated Critical Habitat Likely to be Adversely Affected* (Section 7.2).

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

Effects of the Action (Section 8): Effects of the action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action. These are broken into analyses of exposure, response, and risk, as well as a programmatic analysis as described below for the species and/or critical habitat that are likely to be adversely affected by the action. The species and critical habitat included in this section will be subject to future step-down consultations as details of certain activities become known and as the Navy receives funding, authorization, and/or prepares to carry out these activities. We include a section (8.1) for stressors that are not likely to adversely affect the species and critical habitat that are analyzed further in this Opinion.

Exposure, Response, and Risk Analyses (Section 8.2): In the Risk Analysis, we evaluate the potential adverse effects of the action on ESA-listed species and designated critical habitat under NMFS jurisdiction without consideration of the PDCs. To do this, we begin with problem formulation that identifies and integrates the stressors of the action with the species' status (Section 5) and the Environmental Baseline (Section 7) and formulate risk hypotheses based on the anticipated exposure of listed species and critical habitat to stressors and the likely response of species and habitats to this exposure. Future step-down consultations will further identify the number, age (or life stage), and sex of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong as needed. The effects analysis in step-down consultations will also assess the consequences of the responses of individuals of ESA-listed species that are likely to be exposed to the populations those individuals represent, and the species those populations comprise in more detail as required. We also consider whether the action will result in impacts to the essential physical and biological features (PBFs) and conservation value of designated critical habitat. The programmatic analysis evaluates whether the implementation of the applicable PDCs is sufficient to ensure that the action will not increase the risk to ESA-listed species or the function of the PBFs and conservation value of designated critical habitat associated with the implementation of the proposed action over the consultation lifetime.

Cumulative Effects (Section 9): Cumulative effects are the effects to ESA-listed species and designated critical habitat of future state or private activities that are reasonably certain to occur within the action area (50 C.F.R. §402.02). Effects from future Federal actions that are unrelated to the action are not considered because they require separate ESA section 7 compliance.

Conclusion (Section 10): With full consideration of the status of the species and the designated critical habitat, we consider the effects of the action within the action area on populations or subpopulations and on PBFs when added to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

- Reduce appreciably the likelihood of survival and recovery of ESA-listed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether the action is likely to jeopardize the continued existence of such species; or
- Appreciably diminish the value of designated critical habitat for the conservation of an ESA-listed species, and state our conclusion as to whether the action is likely to destroy or adversely modify designated critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, then we must identify Reasonable and Prudent Alternative(s) to the action, if any, or indicate that to the best of our knowledge there are no reasonable and prudent alternatives (see 50 C.F.R. §402.14(h)(3)).

For a mixed programmatic consultation, an *Incidental Take Statement* (Section 11) is included for those actions where no step down consultation will occur and take of ESA-listed species is reasonably certain to occur. We anticipate that additional ITSs will be issued for step-down formal consultations for those activities reasonably likely to result in incidental take in keeping with the revisions to the regulations specific to ITSs (80 FR 26832, May 11, 2015; ITS rule). The ITS specifies the impact of the take, reasonable and prudent measures to minimize the impact of the take, and terms and conditions to implement the reasonable and prudent measures (ESA section 7 (b)(4); 50 C.F.R. §402.14(i)).

We provide discretionary *Conservation Recommendations* (Section 12) that may be implemented by the action agency (50 C.F.R. §402.14(j)). Finally, we identify the circumstances in which *Reinitiation of Consultation* (Section 14) is required (50 C.F.R. §402.16).

2.1 Evidence Available for the Consultation

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of Google Scholar, literature cited sections of peer reviewed articles, species listing documentation, and reports published by government and private entities. Searches were used to identify information relevant to the potential stressors (underwater investigations using divers and equipment, underwater cleanup activities, vessel transit, and other operations) and responses of ESA-listed species and designated critical habitat. This Opinion is based on our review and analysis of various information sources, including:

- Information submitted by the Navy
- Government reports
- Peer-reviewed scientific literature

These resources were used to identify information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS jurisdiction that may be affected by the action to draw conclusions on risks the action may pose to the continued existence of these species and the value of designated critical habitat for the conservation of ESA-listed species.

3 Description of the Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies. 50 C.F.R. 402.02.

The Navy proposes the removal of suspected MEC items and munitions constituents (MC) from underwater areas throughout UXO 16. MEC and MC may be present in underwater areas surrounding the former NASD and former VNTR because of munitions transport, OB/OD, and firing activities as part of past military training.

3.1 Authorities under which the Action will be Conducted

On February 11, 2005, the former NASD and VNWR on Vieques were placed on the National Priority List (NPL) as the former AFWTA-Vieques, requiring all environmental restoration activities for Navy Installation Restoration sites on Vieques to be conducted under CERCLA unless and until removed from CERCLA authority. The Navy, DOI, U.S. Environmental Protection Agency (EPA), and Puerto Rico Environmental Quality Board (PREQB, now part of the Puerto Rico Department of Natural and Environmental Resources [PRDNER]) executed a Federal Facility Agreement on September 7, 2007, establishing the procedural framework and schedule for implementing CERCLA response actions in Vieques.

3.2 Proposed Activities

Proposed activities within UXO 16 include:

- Location and removal of underwater munitions items from on or beneath the seafloor
- Collection of aquatic samples such as sediment, water, and biota
- Installation and maintenance of structures, such as anchor systems, marker buoys, and floating barriers
- Underwater investigations using remote sensing and testing of new detection technologies
- General boating operation
- Transplantation of coral and seagrass

All of these activities will be conducted during daylight hours.

The following subsections provide details of the activities that will be conducted by the Navy as part of the CERCLA action in UXO 16 in order to locate, evaluate, and potentially remove MEC and material potentially presenting an explosive hazard (MPPEH), which includes MC. These activities will incorporate the appropriate PDCs (Section 3.4.1) to avoid and minimize impacts to ESA-listed species and their designated critical habitat. Many of these PDCs are already part of the standard operating procedures (SOPs) the Navy has developed in coordination with NMFS for investigations of underwater MEC/MPPEH in UXO 16 and other former naval training areas in Puerto Rico. The effects of these activities are considered in this Opinion (Sections 7.1 and 9) to the extent possible and take incidental to the proposed activities is exempted through the ITS issued with this Opinion. Some of the sampling and removal activities may require step-down consultation under this programmatic Opinion, as described further in the subsections below.

3.2.1 Location and Removal

Location and removal activities require the use of vessels as diving and/or equipment platforms.

Identification of MEC/MPPEH may be conducted in advance of any removal activities or may be conducted the same day as removal activities. Surface exposed MEC/MPPEH locations may be identified using previously conducted underwater video, by diver visual inspection, through underwater surveys with remotely operated vehicles (ROVs), by UXO diver inspections with hand-held metal detectors, or other non-intrusive methods. UXO divers may locate subsurface (buried) MEC/MPPEH items in unconsolidated sediment with hand-held metal detectors or using towed electromagnetic detectors. Detected subsurface metal anomalies can include cultural debris, munitions debris, or MEC/MPPEH and require exposure by hand to enable identification. The maximum depth of hand exploration will be to approximately 2 feet (ft) using hand tools such as spades, trowels, or shovels.

In general, when MEC/MPPEH are found, notes and potentially photos and/or videos will be taken of the item, the surrounding habitat, and the presence and proximity of ESA-listed species and designated critical habitat to the item. This information is used to develop a description of the item, consider if and how the item can be removed safely, and, if present, determine how potential effects to ESA-listed species and critical habitat can be avoided or minimized.

At the start of removal activities, the location of each surface or subsurface MEC/MPPEH item will be approached by boat and temporarily marked with a float (small sandbag with a line and Pelican™ Float, or comparable) placed in the water from the vessel. UXO divers using surface-supplied air, SCUBA, or snorkel, depending on water depth and other considerations, will then visually confirm the target item, aided by a hand-held magnetometer. Once the item is confirmed, the UXO divers will move the marker float to the exact location of the item and conduct an initial assessment of the item's type, level of deterioration, level of encrustation, and any other information pertinent to determining whether the item can be removed and the removal method. Underwater ROVs may be used to support diving operations under this and other tasks and underwater operations will be documented in field notes and potentially using underwater video and/or photographs.

After completing an assessment of each item, the UXO divers will exit the water to discuss their assessment with the project team. Scientific divers, escorted by UXO divers, will enter the water and assess habitat conditions in the planned work zone surrounding the item, including the presence and proximity of ESA-listed species and critical habitat, non-listed hard and soft corals, seagrass, and other habitat features. Alternatively, an ROV will be used to collect the information to assess the habitat. Following the assessment, the UXO divers will finalize the item-specific removal details with input from a project scientist.

In general, the location and removal of a single MEC item is expected to last 2 to 4 hours. If there are multiple items in an area, which is expected to be rare but could occur, the duration of removal activities would be longer.

Removed MEC/MPPEH items will be transported by boat to a shoreline access point, and then by land to an area within the former VNTR to be managed in accordance with the practices that have been established for terrestrial munitions response activities on Vieques. The shore-to-land transfer of items will typically occur at an established boat ramp.

There have been cases of items being encrusted along the shoreline and immediately seaward of the shoreline such as in the area of Puerto Ferro, which is along the southern shoreline of the former VNTR. The Navy completed a removal action for these items in coordination with NMFS and does not anticipate the need for similar removal actions in the future, as no other areas with encrusted munitions along the shoreline have been identified. However, if additional areas are identified in the future, a step-down consultation would be required as described in this Opinion and additional PDCs may be developed as part of the step-down consultation in order to ensure removal actions for encrusting munitions are protective of ESA-listed species and designated critical habitat.

3.2.1.1 MEC/MPPEH Removal Methods

Various MEC/MPPEH removal methods may be used. Where practical, ESA-listed and non-listed coral species and/or seagrass growing in the vicinity of items targeted for removal, and considered likely to be impacted by project activities, will be evaluated for possible relocation prior to item removal (including coral growing on the item itself). Relocation of coral or seagrass prior to implementation of item removal would be undertaken only if deemed safe by UXO-qualified personal. If deemed safe, removal of attached or encrusting corals from a specific item may be conducted underwater prior to the item being secured and taken to an onshore location for further explosive hazard management. Any live coral fragments resulting from underwater removal will be recovered and relocated to the extent practicable.

Direct Removal of Items

The UXO divers will remove items determined to be safe to move by hand from the seafloor. The UXO divers will pick up each item and carry each to the water surface of will attach a bridle or place the item in a basket or suitable substitute for UXO personnel on a retrieval boat to lift it to the surface. Large or heavy items may be lifted from the bottom to the surface using an appropriately sized lift bag/balloon attached to the item or boat winch to assist UXO personnel. If a lift bag/balloon is used, UXO personnel would inflate it and guide the item to the surface for retrieval by personnel on the vessel. A boat-mounted winch would typically be used for extremely heavy items and only in areas where the water depth is sufficient to ensure the boat will not be at risk of contacting the seafloor or benthic biota while maneuvering at or around the item.

Remote Removal of Items

UXO personnel will remove items determined not to be safe for removal by hand remotely using a lift bag/balloon or tripod system. UXO divers will attach a bridle or line directly to the item for either method. Floating lines made of polypropylene or suitable substitute will be used to prevent the lines from affecting benthic habitat. A buoy with a line that exceeds the depth of water by approximately 25 percent will be attached directly to each item to help make the location visible to personnel in the vessel.

Once the item has been remotely lifted off the seafloor, a 5-minute wait time will be observed. Once the 5 minutes have elapsed, the team will return to the item's location, assess the assembly, identify new or potential hazards, and take control of the item.

Lift Bag/Balloon: A lift bag/balloon may be used to remotely remove an item from the seafloor. The lift bag/balloon will be a SUBSALVE USA Bomb Recovery System (BRS)-100, or suitable substitute. This system may be used in areas that have about 4 ft or greater water depths and no ESA-listed coral species within approximately 10 ft of the item.

The lift bag/balloon will be attached directly to the item or to an attachment line or bridle already in place around the item. A pull line will be attached to the lift bag/balloon or item and used to pull the attached assembly off the seafloor. The pull line will be made of polypropylene or suitable substitute so it floats, can be seen on the water surface, and does not impact benthic habitat. Once the pull line is attached, the UXO divers will return to the dive boat and all personnel and vessels will transit to outside the exclusion zone (EZ). One of the vessels will pay out the pull line slowly while exiting the EZ.

Tripod: A tripod with an approximate 3-ft-wide base may be placed directly over the item and used to remotely pull the item from the seafloor, particularly in areas that have a water depth of 6 ft or less. When practical, coral or seagrasses growing at and within approximately 3 ft of the planned tripod/item assembly location would be evaluated for possible relocation prior to item removal (including coral growing on the item itself). As with other item removal methods, relocation of coral or seagrass would be undertaken only if determined safe to do so by UXO qualified personnel. The legs of the tripod will be constructed from three pieces of 1.5-in or larger diameter sturdy material, such as steel or aluminum round pipe, or suitable substitute. The connections will be locked into place using a combination of fittings and fasteners. The tripod will be set up to have an approximately vertical line of pull over the item. It will be secured to the seafloor using sand bags, metal weights, or suitable substitute. The item will be lifted via the pulley on the tripod. The pulley will be operated remotely outside the EZ by using a pull line as described above for the lift bag/balloon, or remotely outside the EZ by using a remote control device without a pull line.

Once outside of the EZ, the Senior Unexploded Ordnance Supervisor/Diving Supervisor (SUXOS/DS) will take a head count, and personnel on the boat will visually survey the area for other vessels, sea turtles, and marine mammals. After confirmation there is no sign of vessels, sea turtles, or marine mammals inside the EZ, the SUXOS/DS will give the approval to remotely move the item. Any slack in the line will be pulled into the boat and the pull line will be secured to a boat cleat, Sampson post, or suitable substitute.

3.2.1.2 Non-Intentional Detonation

A non-intentional detonation during the handling of underwater munitions items is considered to be highly unlikely based on terrestrial and underwater activities conducted by the Navy since 2002 (CH2M Hill 2018). If a non-intentional detonation were to occur, it would present a risk to ESA-listed marine mammals, sea turtles, fishes, and corals, and elkhorn and staghorn coral critical habitat due to the acoustic impacts from the detonation, associated sediment resuspension and transport, and potential structural damage from the detonation.

The Navy will identify mitigation zones to be monitored during the removal of underwater MEC items to minimize the potential for impacts to ESA-listed species and critical habitat from a non-intentional detonation. Mitigation zones are areas at the surface of the water within which MEC removal activities will be temporarily ceased or modified to protect specific biological resources from a non-auditory or an auditory injury to the maximum extent practicable.

Mitigation zones will be specifically identified for each MEC item targeted for removal, and will represent the predicted average distance to a permanent threshold shift (PTS) for ESA-listed species in the work area. Based on the net explosive weight and depth of the MEC item, the longest (and therefore most conservative) average distance to onset of PTS for species that are expected to occur in the work zone will be identified as the mitigation zone.

If removal activities target items UXO personnel have determined present a known or suspected significant detonation hazard (versus items UXO personnel have determined are expended materials or present a low risk of detonation), a step-down consultation will be required as described in this Opinion. The step-down consultation will evaluate the proposed mitigation zone and whether additional PDCs and/or an additional ITS are needed for a particular removal activity. In addition, if removal activities result in non-intentional detonations, a step-down consultation or reinitiation of consultation may be necessary in order to determine whether additional PDCs and/or incidental take authorization are required to be protective of ESA-listed species and designated critical habitat in the action area.

3.2.1.3 Blow-in-Place and Encapsulation

The Navy does not anticipate the use of blow-in-place (BIP) or encapsulation of munitions at this time as part of the removal actions. At the time of consultation initiation, the Navy anticipates that items will be removed from the water and taken to an onshore location for further explosive hazard management. If a determination is made to use BIPs or encapsulation to remove or reduce the threat of underwater MEC to human health and safety, a step-down consultation will be conducted as described in this Opinion.

3.2.2 Aquatic Sample Collection

Collection of samples in UXO 16, such as for chemical analysis, may be required to characterize environmental media (surface water, sediment) and/or to support assessments of potential human health or ecological risk, which may include collection of biological samples (e.g., fish and invertebrates). Surface water and sediment sampling is expected to take no more than an hour at each sampling location.

3.2.2.1 Surface Water Sampling

Surface water samples may be collected by boat, either using tubing attached to a peristaltic pump or a water grab sampling device lowered over the side to various water column depths. The *Master Standard Operating Procedures, Protocols, and Plans Revision* (NAVFAC 2017) specifies that surface water samples be collected within the top 12 in of the water column unless a specific depth strata is identified in the sampling and analysis plan for a particular project. Water samples will be collected prior to sediment samples in cases where both water and sediment collection are part of the activity.

In some cases, scientific divers may be used to collect surface water grab samples at specific locations, such as in the immediate vicinity of MEC/MPPEH items. Collection of surface water samples offshore is not planned and, if it does occur, would likely be very infrequent.

3.2.2.2 Sediment Sampling

Sediment sampling will be limited to non-coral areas or sand channels/sand areas within reef areas. Sediment samples will be collected using either a small clamshell-type dredge (e.g., petite Ponar® dredge with a 6-in by 6-in opening, weighing about 25 pounds) lowered from a boat, or by hand using scientific divers, if precision sampling is necessary or in conditions not suitable for the use of a clamshell dredge. The stainless steel, clamshell dredge is designed to collect a 6-in deep sediment sample after lowering it to the sediment surface. Diver collection of sediment may include use of a sediment coring device such as a “push tube” (that is, Lexan tube), or a short stainless steel sediment auger with a Lexan tube insert, to collect 0 to 6-in sediment samples. The tubes are usually 2 to 3-in diameter and 6-in-long. If sediment cannot be adequately collected or retained using a coring technique, such as in areas with very shallow sediment or loose sediment that cannot be retained in the core, sediment will be collected using a spoon or scoop and placed directly into a sample jar. Sediment collection is an activity that may require a safety inspection by munitions response divers using a metal detector to verify there are no MEC at the collection point.

3.2.2.3 Biological Sampling

Biological samples typically include species of fish or invertebrates for use in estimating potential risk to human or ecological consumers. Collection methods will vary depending on species and habitats. Fish and invertebrate sampling can include the use of nets or traps. The *Master Standard Operating Procedures, Protocols, and Plans, Revision - 2017* (CH2M Hill 2017) indicates that gill nets, Fyke nets, seine nets, or cast nets may be used to collect fish and invertebrates.

Because the BA indicates that only cast nets will be used for biological sampling requiring nets, we analyze the effects of the use of cast nets only in this Opinion. If the Navy decides to use one of the other types of nets identified in the master SOP document for biological sampling, reinitiation of consultation may be required depending on the potential effects of the use of other net types on ESA-listed species.

Cast nets, a circular monofilament net with a lead-weighted perimeter, are designed to be thrown opportunistically in areas of schooling fish or in areas likely containing invertebrates such as shrimp or crabs. Cast nets are thrown in areas free of bottom snags because these may tear the net or prevent it from closing properly.

Traps (typically collapsible, polyethylene mesh traps with double openings) would be baited and deployed in non-coral habitats appropriate for the target species, frequently checked for captured organisms (e.g., fish or crabs), and re-baited as necessary. Traps are deployed from a vessel. A rope and float are attached to each trap along with any identification as part of permits (if required). If multiple traps are deployed, they are initially spaced evenly over a sampling area but may be redistributed to target areas with higher catch rates. The global positioning system (GPS) coordinates of each trap are recorded, along with deployment time and approximate water depth, at the time of deployment. To prevent entanglement of ESA-listed species, traps would be deployed in non-coral shallow areas, and would be attended full time and frequently checked while either deployed by divers or using ROVs. Trap lines would be designed to remain taut to minimize entanglement risk for listed species such as sea turtles. Traps may remain in an area for 24 hours.

Scientific divers may be used to collect target fish and invertebrate species that are not readily captured using the above techniques, or when habitat conditions (e.g., presence of coral reefs, strong currents) preclude the use of nets or traps. Divers would collect specimens directly by hand, with either small nets or using other non-invasive methods.

Collection of coral tissue samples could be a requirement of future investigations, for example, to evaluate accumulated contaminants. As a result, there is the potential for collecting tissue samples from a total of 50 ESA-listed coral colonies over the estimated 20-year period covered by this consultation. Coring of encrusting corals or fragmentation of branching corals to collect a sample would be done preferentially on large colonies. The BA did not identify whether particular species would be targeted for this sampling or provide specifics of the methodology of sample collection. If coring or other sample collection from ESA-listed coral colonies is proposed in the future as part of biological sample collection, a step-down consultation would be required and additional PDCs may be developed and/or additional incidental take may be authorized.

3.2.3 Installation and Maintenance of Structures

Installation and maintenance of in-water structures such as anchor systems, floating waterway barriers, marker and warning buoys, Autonomous Reef Monitoring System (ARMS), limestone coral attachment plates, Coral Arks, and associated materials like buoy tackle, may be required. These structures are generally intended to support boat anchoring and notification or protection of areas within UXO 16 that may pose a hazard to the public due to the potential presence of MEC/MPPEH.

Anchoring system options will be selected based on the intended use and substrate type. Anchor types that will be used in UXO 16 are described further below. Anchor point locations will not contain live or dead coral and live or dead coral will not be within the swing radius of the anchor chain.

Bulk-type anchors are intended to serve as primary anchors for large pieces of equipment, such as waterway barriers, mariner warning buoys, or boats. They are constructed of concrete with a footprint of approximately 8 ft by 8 ft (64 square feet [ft^2]) and weigh approximately three tons. Bulk anchors will be placed on bare unconsolidated substrates.

Helical (sand screw) and Manta Ray™-type anchors are intended to be connected to bulk anchors or to support smaller equipment types such as marker buoys and are placed in sandy bottom areas. Helical anchors are suitable for sand substrates greater than about 9 ft in depth, have a diameter of 10 in, and have a total footprint of approximately 78 square inches (in^2). Manta Ray™ anchors are suitable for sand depths between approximately 3 and 9 ft, and have a footprint of approximately 60 in^2 from the 5-in by 12-in anchor.

Rock (pin)-type anchors are designed for installation directly into hard bottom substrate, often in areas with corals. These anchors are usually connected to bulk anchors or used to support smaller equipment such as reef marker buoys. Pin-type anchors are installed by drilling or coring holes with a diameter of approximately 4-in and then installing an approximately 18-in-long galvanized metal pin with a large eye for connecting shackles for attaching the anchor to whatever structure it is supporting. The estimated footprint disturbed by a pin anchor is 28 in^2.

Waterway barriers and buoys are typically attached to anchor systems with stainless steel chain. Waterway barriers that may be used in the future are likely to have the same general characteristics as those installed in the area of Bahía Icacos on the northeast of Vieques. Those barriers, which were the subject of an ESA section 7 consultation concluded in 2012 (NMFS 2012), were module with each module having a 120-in length, 16-in diameter, and weight of 100 pounds. The modules have an adjustable draft between 3 and 8-in.

In some instances, an underwater buoy is attached midway along the anchor chain of buoys and barriers to keep any slack in the chain from abrading the seafloor and to serve as a backup float should the surface buoy detach.

As part of an ESTCP funded demonstration project to develop a new coral reef conservation technology, identified as Coral Reef Arks, NIWC Pacific has proposed to deploy ARMS, limestone coral attachment plates, and Coral Arks in UXO 16.

A Coral Reef Ark is a geodesic, slightly positively buoyant structure approximately 4-ft in diameter that would be anchored to the seafloor in suitable environmental conditions and would serve as the platform onto which multiple ARMS and limestone coral attachment plates would be fastened. Coral Reef Arks are intended to be installed about one year after ARMS and coral attachment plates have been deployed in UXO 16 reef habitats.

Individual ARMS consist of a stack of 9, narrowly spaced 9 x 9-in square PVC tiles affixed with stainless steel bolts to a larger PVC baseplate (14 x 18 in). ARMS are intended to be placed on the seafloor near living corals for at least 1-year, thus becoming seeded naturally with coral reef organisms. A total of 40 ARMS are proposed to be deployed near the western end of Vieques, all within UXO 16 in the vicinity of SWMU 4. Specific locations for ARMS placement within these areas will be identified by scientific divers escorted by UXO divers. ARMS will not be placed on critical habitat for staghorn/elkhorn corals. Either hardbottom habitat covered with turf algae (where coral larvae would not settle), rubble accumulation areas, or sand bottom habitats will be selected for ARMS placement. Multiple ARMS units (at least five) will be fastened to each other, typically in a linear configuration, weighted at either end with hardened bags of concrete (50 - 80 pounds each), and placed on the seafloor. Locations will be selected that do not have ESA-listed corals in the immediate vicinity should the ARMS assemblies or concrete anchors move during a storm event.

In addition, during the MEC/MPPEH removal action previously described, some non-listed hard and soft corals, and potentially some ESA listed corals, requiring transplantation to nearby natural substrate to prevent loss or damage may instead be attached to limestone plates for future transfer to a Coral Ark (after approximately 1 year). Each plate is composed of an 8 x 8-in square limestone tile glued to a 9 x 9-in square PVC tile. These limestone plates will either be affixed to hardened bags of concrete in appropriate habitat (as described above for ARMS) or attached with cable ties to suitable rocky substrate that is not critical habitat and does not contain ESA-listed corals.

The design of the Coral Ark is currently conceptual. Construction materials, anchoring system, deployment locations, and a long-term monitoring program (structural and biological) have yet to be determined. Once the Ark design, placement, and other details have been determined, the Navy will submit project-specific information (per the requirements described in Section 3.3.2. This information will be used in a step-down consultation in order to analyze the effects of the Arks and to determine whether additional PDCs and/or incidental take authorization are required to be protective of ESA-listed species and designated critical habitat in the action area.

The monitoring/maintenance schedule for in-water structures will be flexible to account for observations of structural component integrity made during previous monitoring/maintenance activities, sea and weather conditions since the previous activity, etc. This will likely result in monitoring/maintenance events being conducted on an approximately monthly or bimonthly cycle, but the frequency will be adjusted as noted. Surface components are typically inspected from a boat and underwater components are inspected with an ROV. Divers will be used when detailed inspection, preventive maintenance, or underwater repair is needed. Activities done from a vessel or in-water by divers include removal of biofouling organisms; inspection/replacement of underwater chain, linkages, or sacrificial anodes; and inspection/replacement of primary or secondary underwater flotation. Maintenance activities at each anchor location are expected to take 2 to 4 hours.

It may be necessary to transplant ESA-listed corals and/or seagrass from the proposed footprint of in-water structures, particularly anchors, if the most suitable location for these structures (such as marker buoys) is an area containing these resources. If transplant of coral or seagrass is required as part of the installation of in-water structures in the future, a step-down consultation would be required and additional PDCs and/or incidental take authorization may be required.Underwater Investigations

3.2.4 Underwater Investigations

A WAA DGM survey of UXO 16 was completed in 2017, but future focused remote sensing surveys may be required to pinpoint item location or to check whether items have moved during storms, among other reasons. Underwater DGM or other remote sensing technologies may be required to investigate specific locations or areas for potential individual munitions, or areas with possible high densities of metallic anomalies.

The categories of remote sensing technologies that may be used include:

- Traditional manned approaches – These are diver-conducted seafloor surveys typically done using a hand-held metal detector. This method has been applied in UXO 16 at localized sites to avoid contact with munitions during intrusive activities (e.g., installation of anchors at Bahia Icacos, sediment sampling in UXO 16 adjacent to SWMU 4), and for MEC/MPPEH surveys.
- Marine towed array – A boat-towed marine magnetometer array is commonly comprised of a high-density foam wing with a fiberglass exterior equipped with multiple Cesium-vapor magnetometers and/or electromagnetic induction sensors to detect the presence of ferrous metallic objects. The wing is towed behind an A-frame mounted to a survey boat, such as that used for the WAA in 2017. An array is usually equipped with weights, an altimeter, pressure sensors, digital compass to record boat pitch and roll data, a top-side GPS unit for real-time positioning information, and recorders to document distance and horizontal angle at which the array was being towed. The wing used for the WAA had weights attached to the wing keels and to the wing tips as ballast so the wing would fly level while towed. An adjustable weight system was mounted on the towline in front of the wing to help control survey depth. An underwater video system, such as forward and rear-facing cameras, is often also attached for live-feed monitoring of seafloor conditions,

including benthic habitats/species to be avoided, as well as identifying objects that are potential munitions. During the WAA, a GoPro™ HERO4 Black equipped with a high-capacity battery was mounted in front of the array and recorded in high-definition at 60 frames per second to allow for generally clear images when paused during data review. A high-definition drop camera that provided real-time video information to the survey boat, also recorded video as a backup for the GoPro™.

- Amphibious platforms –
  - ROV-Sensor Integration is used on small to large ROVs that are outfitted with various metal-detecting sensors for bottom surveying. The equipment is typically tethered to a vessel and includes positioning sensors, live-feed video monitoring, and thrust controls for stability and maneuverability.
  - Bottom Crawler-Based Sensing uses amphibious bottom crawlers that include large, motorized, wheeled or tracked vehicles that move along the seafloor and tow a platform with various metal-detecting sensors. This platform would only be used in unconsolidated sediment areas (e.g., sand or mud), and would not be used in habitats such as seagrass or reefs.

The use of hand-held equipment by divers, towed arrays, and ROVs allow operators to monitor and adjust the depth of the equipment to account for changes in sea conditions, bottom topography, and presence of marine organisms while maintaining equipment near the seafloor for better detection of possible munitions items.

Specific remote sensing technologies to be used will be based on the objectives of any future required survey and the Navy assumes these would fall into the same categories as the activities identified in this section. If a determination is made that additional remote sensing investigations are needed that involve technologies that differ substantially from those identified in this Opinion, a step-down consultation will be conducted as described in this Opinion. Similarly, if there are additional projects to test technology or perform other underwater investigations or demonstration projects that are not specified in this Opinion, a step-down or individual consultation, depending on whether the new projects fall within the scope of this consultation, will be required.

3.2.5 Boating Operations

Boating operations are required to support the other activities described in this Opinion, as well as for visually inspecting beach conditions, transporting equipment or personnel, or monitoring EZs during terrestrial detonations of MEC/MPPEH that are part of the CERCLA activities on Vieques.

Boats typically used for water operations include, but may not be limited to the following:

- Rigid-hulled inflatable boat (RHIB) – 21-ft length, single outboard engine
- Zodiac M-470 – 16-ft length inflatable, single outboard engine
- Express boat – 32-ft length, inboard engine

- Mako – 28-ft length, twin outboard engines
- Boston Whaler – 38-ft length, 4 outboard engines

Marine access points within UXO 16 that may be used include:

- Mosquito Pier boat ramp
- Bahia Icacos boat ramp
- Laguna la Plata boat ramp
- Playa Jalova boat ramp

When boats are not in use, they will be hauled out of the water daily (smaller boats) or tied to existing moorings or have temporary moorings installed for task-specific purposes using one of the smaller anchor systems described in Section 3.2.3.

Vessel transit from ports on Vieques outside UXO 16 and the main island of Puerto Rico to areas within UXO 16 where investigation, removal, and other activities considered in this Opinion are taking place is also considered part of the proposed action in this Opinion.

If temporary or permanent boating access ramps in locations other than the four access points listed above or improvements to the marine access points listed above are required to support the proposed underwater activities in UXO 16 described in Section 3.2, reinitiation of consultation or a step-down consultation may be required in accordance with the procedures described in this Opinion in order to determine whether additional PDCs and/or incidental take authorization will be needed.

3.2.6 Transplantation of Coral and Seagrass Due to Munitions Removal

Coral relocation in conjunction with munitions removal will be performed to the extent practicable. (The Navy has estimated the potential number of ESA-listed corals on or adjacent to potential MEC/MPPEH as discussed in later sections of this Opinion.) UXO-qualified personnel will determine whether coral adjacent to or attached to a MEC item is safe to remove. If safe, a scientific diver will remove the coral under the supervision of UXO personnel; otherwise, the UXO personnel may be required to perform the coral removal while following instructions from the scientific diver. If coral colonies can be safely removed from a munitions item to which they are attached, these corals will be transplanted from the munition to the site that was occupied by the munition. If corals cannot be reattached at the munitions removal site, they may be transported to locations having habitat conditions similar to the removal site, or otherwise suitable for the species being transplanted. Location conditions to be considered include general health of existing wild populations of corals (e.g., no obvious bleaching or prevalence of diseases), suitable water depth, optimal bottom type (i.e., hard bottom), good water quality (e.g., constant water flow, good light penetration), and limited biological stressors (e.g., coral predators and benthic space competitors such as algae, sponges gorgonians, and fire corals). In addition to ESA-listed corals, non-ESA-listed hard and soft corals that are likely to be damaged or destroyed because of the removal action will also be considered for relocation.

To the extent possible, coral relocations will be conducted the same day as their removal. Removed coral specimens will be temporarily held in separate containers (e.g., plastic buckets) to prevent colonies from contacting each other, kept submerged in water, and held in protected conditions (e.g., temporarily staged underwater in open or vented containers near the removal site for quick re-attachment following item removal or in a cooler or in shaded conditions on the support boat).

Before transplanting, all fouling organisms and sediment will be cleared from the substrate using wire brushes or scrapers. Materials used to secure corals will consider the coral species, size of the coral transplant, substrate characteristics, and typical current or wave energy in the area. The most common attachment materials are 2-part epoxy, hardened masonry nails, and nylon cable ties or coated wires; and Portland cement. Using masonry nails and cable ties is a good method for attaching branching corals, while Portland cement is the best option for large boulder corals.

Relocation-specific information will be collected at the time of transplantation including the GPS coordinates of transplanted ESA-listed corals. Individual colonies or colony clusters will also be field marked using a nylon cable tie with a number-coded “cattle tag” attached to a nominal 3-in hardened masonry nail driven into the substrate near the transplant(s). Encrusting growth on the tags can be scraped off to reveal the number, as necessary, and the metal nail may be relocated using a metal detector, if necessary. Photographs of transplanted colonies with a ruler or other object showing the size of the colony will be taken at the time of transplant. A map of all transplanted ESA-listed corals will be maintained as transplants are conducted.

Success monitoring may be conducted when divers are near transplanted corals during subsequent munitions removal activities. Inspections may be conducted using an ROV or by a scientific diver. Inspections will include, to the extent possible, documentation (including photos) of colony size and condition such as healthy and growing, partial or complete mortality, presence of disease, significant damage from coral predators (corallivores) such as fish, snails, or other invertebrates, and overgrowth or encrustation by organisms such as algae, sponges, tunicates, and cnidarians.

If transplant of ESA-listed corals to man-made materials is proposed, such as to Coral Arks (Section 3.2.3), a step-down consultation will be required as described in this Opinion.

Location and removal of surface and subsurface munitions may affect seagrass. Following a removal from seagrass habitat, a qualified person (e.g., scientific diver) will inspect the location and determine the type of seagrass restoration measures, if necessary, that should be implemented. Qualified personnel (e.g., scientific divers) with experience in seagrass restoration techniques will conduct all seagrass restoration. Any void created on the seafloor by an inadvertent impact will be backfilled with adjacent sediment so the grade of the impacted area is approximately flush with the surrounding grade.

The methods used to restore seagrass will be specific to the condition of the impacted seagrass and the seagrass species affected. Displaced rhizome segments or small seagrass plugs will be re-planted by hand, using biodegradable pins, if necessary. In instances where larger subsurface items are being investigated, an area of seagrass can be cut on three sides and rolled up to allow better access to the anomaly. Afterwards, the excavated area will be backfilled with the removed substrate and the seagrass rolled back into place and pinned (for plugs greater than approximately 8-in across) with biodegradable stakes. Small areas of disturbance are expected to backfill and recolonize naturally so not all work done in seagrass will include restoration.

3.3 Programmatic Consultation Requirements and Procedures

This section details the non-discretionary PDCs that describe aspects of the action required for activities implemented as part of the Navy's cleanup activities under CERCLA in UXO 16 around Vieques Island to avoid or minimize adverse effects on ESA-listed species and designated critical habitat. The section also describes the procedures for streamlined project-specific review and for step-down consultations. Finally, the section details the periodic comprehensive review procedures for the program.

The following additional elements of programmatic consultations are covered in later sections of the Opinion:

- Description of the manner in which activities to be implemented under the programmatic consultation may affect listed species and critical habitat, and evaluation of expected level of effects from covered activities (Sections 7.1, 9, and 10).
- Process for the evaluation of the aggregate or net additive effects of all activities expected to be implemented under the programmatic consultation (Section 9).
- Procedures for tracking and monitoring projects and validating effects predictions, in addition to those contained in this section of the Opinion, are also found in the Incidental Take Statement, including its RPMs and associated terms and conditions (Section 13).

The proposed programmatic action includes specific activities that are (1) not likely to adversely affect ESA-listed species and their designated critical habitat with implementation of applicable PDCs, and (2) are likely to adversely affect ESA-listed species and their designated critical habitat, even with implementation of PDCs. While some activities have ESA section 7 determinations made under this programmatic opinion, there are others that are likely to adversely affect ESA-listed species and their designated critical habitat that will require a step-down consultation. For activities that may result in take of ESA-listed species, additional RPMs to reduce or minimize the effect of the take may be developed as part of a step-down consultation. Although some PDCs and RPMs appear similar, the implementing terms and conditions of the RPMs provide specific, non-discretionary requirements that the action agency must follow.

3.3.1 Project Design Criteria

The Navy has developed SOPs for underwater surveys and removal actions around Vieques as part of past ESA section 7 consultations and as part of the on-going coordination between the

Navy and NMFS as the Navy works to meet CERCLA requirements while also complying with the ESA.

PDCs have been identified to limit environmental effects of location and removal of items, aquatic sample collection, installation and maintenance of structures, underwater investigations, boating operations, and transplantation of coral and seagrass associated with MEC removal described in Section 3.2, and vessel transit, described in Section 4. Some of the PDCs related to location and removal of items are meant to reduce the possibility for a non-intentional detonation to occur, but a step-down consultation will be required for removal of items suspected to present a significant detonation hazard as described in Section 3.2.

PDCs for BIPs and encapsulation have not been included in this Opinion as the Navy does not anticipate using these methods at this time and step-down consultation would be required if these methods are used in the future. Should BIP and/or encapsulation become part of the in the future, step-down consultations for these activities would be required and additional PDCs may be developed and/or incidental take authorized as part of these step-down consultations.

PDCs have been included for the use of cast nets to collect biological samples and for the collection of coral tissue samples. However, the use of nets other than cast nets and specific coral tissue sampling would require step-down consultation that may include the development of additional PDCs and/or incidental take authorization. Similarly, the use of remote sensing technologies that are different from those described in this Opinion, the transplant of seagrass and/or coral as part of the installation of in-water structures, and the creation of new temporary or permanent boat access points or improvements to existing boat ramps for future in-water activities in UXO 16 may require step-down consultation and the development of additional PDCs and/or the authorization of additional incidental take.

The PDCs included in this Opinion are taken from the SOPs the Navy implements during underwater cleanup activities around Vieques and other former naval sites in Puerto Rico and the conservation measures the Navy included in the BA for this consultation. The PDCs also include additional requirements NMFS believes are necessary to avoid and minimize potential adverse effects of the action on ESA-listed species and designated critical habitat based on consultations involving underwater MEC, vessel operations, and installation of buoys and other in-water structures. These PDCs, when applied to in-water activities associated with the Navy's CERCLA activities in UXO 16, minimize the negative effects of these activities to ESA-listed species and designated critical habitat.

<u>General PDCs applicable to all activities addressed in this consultation</u>:

1. Prior to initiating on-water work, field personnel will receive training or briefings, as applicable, regarding the potential presence of threatened or endangered species that may be encountered, their physical characteristics, preferred habitats, how they can be identified, actions to be taken if sighted, and avoidance measures to be followed as

detailed in the PDCs in this Opinion. This training or briefing will be prepared and offered by qualified personnel (e.g., biologist, marine biologist, environmental scientist).

2. Personnel will be advised that there are civil and criminal penalties for harming, harassing, killing, or otherwise altering the natural behavior or condition of threatened or endangered species protected under the ESA.
3. A log detailing endangered or threatened species sightings in marine habitats will be maintained during implementation of the activities within UXO 16 described in this Opinion. The log shall include, but not be limited to, the following information: date and time, location coordinates using a GPS unit, species identification, behavior of the animals, one or more photographs (if possible), and any actions taken because of the sighting during the work period. Copies of the logs will be submitted to NMFS Office of Protected Resources Interagency Cooperation Division as part of the annual reporting requirements.
4. Each team performing intrusive underwater investigation work will be accompanied by qualified and experienced personnel (e.g., biologist, marine biologist, environmental scientist, among others) in order to identify the presence or absence of threatened or endangered species in the work area and direct avoidance measures as needed.

<u>PDCs applicable to boating operations</u>:

1. All vessel operations will take place during daylight hours.
2. Vessel operators shall use caution, be alert, maintain a vigilant lookout and reduce speeds, as appropriate, to avoid collisions with marine mammals, sea turtles, and ESA-listed fish (particularly elasmobranchs) and to avoid accidental groundings during the course of normal operations.
3. During vessel operations, when marine mammals, sea turtles, or ESA-listed fish (particularly elasmobranchs) are sighted or known to be in the immediate vicinity, operators are required to employ all possible precautions to avoid interactions or collisions with animals, including the following:
    a. Reducing speed
    b. Avoiding sudden changes in speed and direction, or if a swimming marine mammal, sea turtle, or large ESA-listed fish species is spotted, attempting to parallel the course and speed of the animal so as to avoid crossing its path
    c. Avoiding approach of sighted animals head-on or from directly behind.
    d. When whales are sighted, maintain a distance of 100 yards (yd; 91 m) or greater between the whale and the vessel.

e. When sea turtles or dolphins are sighted, attempt to maintain a distance of 50 yd (46 m) or greater between the animal and the vessel wherever possible. Try to maintain a distance of 50 yd from ESA-listed fish as well (particularly elasmobranchs).

f. Sea turtles and marine mammals may surface in unpredictable locations or approach slowly moving vessels. When an animal is sighted in the vessel's path or in close proximity to a moving vessel, reduce speed and shift the engine to neutral. Do not reengage the engines until the animal is clear of the area.

4. Reduce vessel speed to 10 knots or less when mother/calf pairs, groups, or large assemblages of whales are sighted near an underway boat, when safety permits. A single whale at the surface may indicate the presence of submerged animals in the vicinity. The boat should attempt to route around the animals, maintaining a minimum distance of 100 yards whenever possible.

5. Watercraft will travel at no wake speed within shallow waters 10 ft or less and/or when 150 ft from the coastline.

6. While on station, work areas will be routinely monitored for the presence of sea turtles, marine mammals, and ESA-listed fish species (particularly elasmobranchs) at/near the water surface during boating or surface operations and below water by video or divers when in-water work is being conducted.

7. Any collision with and/or injury to a marine mammal or sea turtle shall be reported immediately to the appropriate NMFS office and local authorized stranding/rescue response organizations (see https://www.fisheries.noaa.gov/report for regional contact information for reporting). Work personnel should report sightings of any injured or dead sea turtle or marine mammal immediately to NMFS and the PRDNER regardless of whether the injury/death is caused by the Navy's activities. Collisions with ESA-listed fish or coral should also be reported to NMFS and PRDNER.

8. If injury or death of an ESA-listed species is caused by a boat collision or other work activity associated with the action described in this Opinion, the work personnel involved in the activity will remain available to assist the response personnel as needed.

9. When planning transit routes, deep-water routes will be preferentially selected where possible.

10. Vessel operators will review nautical charts and use onboard depth sounders to prevent boat contact with the seafloor and coral colonies that extend toward the surface.

11. Vessels will be anchored preferentially in unvegetated sandy bottom whenever possible. If anchoring in unvegetated sandy bottom is not possible, vessels may anchor in vegetated bottom with seagrass and/or algae. Vessels will not anchor on hard bottom that contains hard and/or soft corals, regardless of the percent coral cover present. The type of

bottom will be confirmed by divers, onboard using a glass-bottom bucket, or by other appropriate means prior to anchoring.

12. If a vessel is anchored in vegetated bottom, the anchor will be removed from the seafloor in a manner that minimizes disturbance to the vegetation; for example by attaching a secondary anchor line to the rear of any plow-type anchor (Danforth®, Union, claw anchor) and pulling the anchor free from the seafloor before lifting it to the surface.

PDCs applicable to location and removal:

1. All underwater and above-water activity will occur during daylight hours.
2. The operational area will be routinely monitored by onboard personnel for the presence of sea turtles and marine mammals, and underwater personnel for the presence of ESA-listed fish. If an animal is observed in close proximity to underwater activities, divers will stand by until the sea turtle or marine mammal moves away from the immediate work area to a point where it cannot be directly contacted by divers or equipment. Should the animal not show signs of leaving, the diver team will leave the location and return to complete the work later. No animals will be chased.
3. If a lift bag/balloon is used for items that cannot be removed by hand, UXO personnel would inflate it and guide the item to the surface for retrieval by personnel on the vessel. All operations will be conducted in a way that will minimize contact with the seafloor and surrounding benthic organisms, including ESA-listed corals.
4. A lift bag/balloon will only be used in areas that have 4 ft or greater water depths and no ESA-listed coral species within approximately 10 ft of the item to be removed.
5. If a boat-mounted winch is used for extremely heavy items, it will only be used in areas where the water depth is sufficient to ensure the boat will not be at risk of contacting the seafloor or benthic biota while maneuvering at or around the item.
6. If a tripod assembly is used to remotely remove items from the seafloor, when practical and determined safe to do so by UXO qualified personnel, coral or seagrasses growing at and within approximately 3 ft of the planned tripod/item assembly location would be evaluated for possible relocation prior to item removal (including coral growing on the item itself).
7. If a tripod is used, it will be secured to the seafloor using sand bags, metal weights, or suitable substitute to minimize the potential for it to move during removal operations.
8. Floating lines made of polypropylene or suitable substitute will be used during removal actions with lift bags/balloons or tripods to prevent the lines from affecting benthic habitat.
9. A buoy will be attached directly to each item to help make the location visible to personnel in the vessel, which will also enable personnel to minimize the potential for

entanglement of towlines in benthic habitats or with swimming animals such as sea turtles.

10. During peak nesting of leatherback (April to July), hawksbill (June to November), and green (August to October) sea turtles, prior to any removal actions involving suspected MEC, surveys of the area within the calculated mitigation zone (see Section 3.2.1.2) will be conducted to look for these animals because of the risk of acoustic effects from a non-intentional detonation. If any of these animals are sighted, work will be delayed until the animals have not been seen for 30 minutes within the mitigation zone. If these surveys are not conducted, reinitiation or step-down consultation will be required to determine whether additional PDCs are needed and/or whether take will occur, the effects of which need to be analyzed.

<u>PDCs applicable to aquatic sample collection</u>:

1. Nets and traps for sampling fish and invertebrates will not be deployed in coral habitats.
2. Cast nets are the only type of nets proposed for use at this time. The use of other types of nets will require a step-down consultation prior to their use.
3. Cast nets will not be deployed in an area where a sea turtle is observed surfacing to breathe.
4. Nets and traps used for sampling marine fish and invertebrates will be attended at all times while deployed.
5. Lines attached to traps will be kept taut and traps will be checked frequently (at approximately 15-minute intervals) to ensure no juvenile sea turtles or Nassau grouper have entered a trap and that no sea turtles or marine mammals are active in close proximity to a trap to minimize the potential for entanglement. If sea turtles or marine mammals are active in close proximity to a trap, it will be temporarily pulled to avoid possible entanglement. Entanglement of sea turtles may require reinitiation of consultation.
6. All sampling equipment that is lowered into the water column will be visually monitored from the boat with either an underwater camera or other type of underwater viewer, or in the water by snorkelers or divers.
7. Divers collecting samples will deliberately avoid sampling in close proximity to ESA-listed coral colonies where unintentional contact could occur.
8. Sediment sampling will generally be limited to non-coral areas or sand channels within reef areas where sufficient unconsolidated sediment for sampling can be found.
9. If sediment samples are collected from habitats containing seagrass, scientific divers will restore disturbed or uprooted plants following the PDCs for transplant of seagrass (below).

10. ESA-listed corals will be avoided during diver hand sampling of fish and invertebrates unless sample collection is targeting coral tissue.
11. Multiple coral cores or coral fragments for coral tissue analysis will not be collected from the same colony. If the collection of multiple samples from an ESA-listed coral colony is proposed, a step-down consultation will be required. Other requirements for the collection of tissue samples from ESA-listed corals authorized through this Opinion are detailed in Section 12.
12. The tools used to collect coral tissue samples will be sterilized between sample collections such that the tools are never used on multiple coral colonies in order to minimize the potential spread of disease.
13. Coral sample collection from massive coral colonies will be done preferentially on large colonies using sterilized corers designed for coral coring. Holes left by coring will be filled with Portland cement or clay immediately following coring in order to minimize the susceptibility to disease and encourage regrowth over the impacted area.
14. Coral sample collection from branching coral colonies will be done preferentially by collecting fragments that have broken from larger colonies naturally. If natural fragments are not available from branching coral colonies, samples will be collected from the outermost portion of the branching tip using sterilized shears or pliers.

<u>PDCs applicable to installation and maintenance of structures</u>:

1. Most anchor systems will only be installed in sand or mud substrates and where ESA-listed corals and critical habitat are beyond the reach of the anchor chain in the event of a surface buoy failure.
2. Seagrass habitat will be avoided to the extent possible for anchor installation. If anchors have to be installed in seagrass, a location with minimum seagrass cover will be identified for anchor installation. Subsurface buoys will be installed to keep any chain slack from impacting seagrass.
3. New anchor points for sand screws will be located where there will be the least potential for environmental impacts while allowing marker buoys to be securely anchored and in a location where they will be effective in terms of being readily viewed by boaters.
4. Anchor point locations must not contain live or dead coral and live or dead coral must not be located within the potential reach of the anchor chain (i.e., live or dead coral must not be within 3 meters [m] of the estimated swing radius of the chain).
5. Sand screws will be preferentially located in deep unconsolidated sediment with limited biological cover of macroalgae and/or seagrass.

6. Seagrass disturbed or displaced (such as for bulk anchor installation) will be transplanted by qualified personnel as applicable following the PDCs for seagrass transplant (below).
7. In locations where marker buoys will be anchored in hard substrate, the anchor location must be bare rock or rock covered with macroalgae with no live or dead coral. Pin anchors will be used in hard substrate in areas where existing ESA-listed corals are beyond the reach of any attached chains or equipment. A subsurface buoy will be attached along the anchor chain to prevent scouring of hard bottom habitat or damage to future coral recruits.
8. A dedicated observer will be present on work vessels to look for sea turtles and marine mammals. If a sea turtle or marine mammal is observed in close proximity to maintenance activities, work will stop until the animal moves out of the work area of its own volition.
9. If it is determined that modifications to a waterway barrier or other in-water structure, including specific types of system components and final system design or types of anchors to be used, are necessary at the time of installation, NMFS will be notified of these modifications prior to installation. Modifications that increase the type or extent of adverse effects evaluated in this Opinion may require a step-down consultation or reinitiation of consultation.
10. Turbidity will be visually monitored underwater during all construction activities. In the event that sediment plumes are generated because of the activity, all construction activity will cease and measures to reduce turbidity will be implemented.
11. If structures are installed during sea turtle nesting season, beaches in the project area will be monitored daily for signs of nesting activity.
12. If structures such as barriers are installed offshore of documented sea turtle nesting beaches, beach monitoring during nesting season will be conducted each year the structure is in the water. Monitoring of the structure during nesting season will also be conducted to determine whether the structure is acting as a barrier to movement of hatchlings.
13. If in-water structures such as waterway barriers are found to affect the movement of sea turtle hatchlings to open water, hatchlings will be transported seaward of the barriers. This transport of hatchlings is being authorized through this Opinion as a reasonable and prudent measure to reduce the impact of the take that would occur in the absence of the measure (i.e., hatchlings blocked by barrier, likely high rate of mortality).
14. ROV inspection of in-water structures will be done by an experienced, qualified person capable of maintaining a safe distance from ESA-listed coral colonies that may be in the area.

15. Diver inspections and repairs of in-water structures will be conducted by qualified personnel who will avoid ESA-listed corals that may be in the area.
16. If helical anchors need to be removed or replaced, these can be turned out of the sediment without damaging the habitat. Bulk anchors, Manta Ray™ anchors, and pin anchors will be left in place because removal activities are likely to result in more damage than simply maintaining these anchors at their original location. However, bulk anchors that are no longer in use will be periodically inspected to be sure they are not causing damage to surrounding habitat. If damage due to the presence of these anchors is observed, a step-down consultation will be required to evaluate these effects.
17. Coral recruits observed on bulk or other types of anchors will be left undisturbed.
18. Coral recruits on chains or buoys, which must be maintained and eventually removed from the water, will be removed and transplanted as feasible.

<u>PDCs applicable to underwater investigations</u>:

1. During remote sensing surveys involving a towed platform, boats will travel between 2 and 3 knots and self-propelled equipment such as ROVs will operate at similar speeds.
2. A dedicated observer will be present to look for sea turtles and marine mammals. If an animal is observed in close proximity to towed arrays or self-propelled equipment, the equipment will be brought back to the work vessel until the animal has exited to area of its own accord and has not been seen for 30 minutes.
3. Vessel operators will use nautical charts, data from previous surveys in UXO 16, onboard depth sounders, real-time video, and sensors on any towed array or self-propelled equipment to prevent the vessel, towed array, and/or self-propelled equipment from contacting the seafloor and underwater obstacles, including coral colonies.
4. Surveys will be conducted in water depths greater than 4 ft. At this depth, the draft of survey vessels, including the propeller, will have at least one foot of clearance from the marine bottom or the tops of coral colonies. Towed arrays and self-propelled equipment will operate at the water surface if the water depth is 4 ft.
5. Because the towed array is naturally buoyant, a counterweight is necessary to help stabilize it at the desired tow depth. The counterweight will be attached to the towing line several feet ahead of the towed array in a manner that prevents it from hanging down from the towing line to eliminate the potential for the counterweight to contact the seafloor or obstacles such as coral colonies. Weight may also be added to the towed array itself to partially overcome its natural buoyancy.
6. A forward-facing video camera with real-time feed to the surface will be mounted to a towed array or a self-propelled ROV. An operator will monitor the video feed at all times during the survey to ensure the equipment is operating at the desired elevation above the seafloor and that contact with the seafloor and any obstacles (including corals)

is avoided. A bow-mounted video camera with real-time feed to the surface may also be employed if it is determined that it would assist in ensuring equipment avoids potential contact with the seafloor or collisions with obstacles, including corals.

7. The auto-winch attached to the towed array will be equipped with a manual override that allows the operator to raise the towed array based on observations made using real-time video rather than waiting for the winch to automatically adjust to changing seafloor elevations.

<u>PDCs applicable to transplant of coral and seagrass</u>:

1. A qualified person (e.g., experienced scientific diver) will inspect all munitions removal locations prior to any removal activities to determine the presence and proximity of ESA-listed corals and critical habitat, and will relay this information, as well as required avoidance procedures, to the dive team.
2. All underwater work personnel will be familiar with the identification of ESA-listed coral species and elkhorn and staghorn coral critical habitat, and the procedures to be followed to prevent impacts to these species or habitats during work activities.
3. If coral colonies can be safely removed from a munitions item to which they are attached, these corals will be transplanted from the munition to the site that was occupied by the munition. The transplant of ESA-listed coral colonies is being authorized through this Opinion as a reasonable and prudent measure that would reduce the take that would occur in the absence of the measure (i.e., coral colonies being damaged or destroyed due to their presence on items to be removed from the water).
4. If corals cannot be reattached at the munitions removal site, they may be transported to locations having habitat conditions similar to the removal site, or otherwise suitable for the species being transplanted. These sites will be selected in coordination with NMFS. Location conditions to be considered when corals have to be transplanted to a new location include: general health of existing wild populations of corals (e.g., no obvious bleaching or prevalence of diseases), suitable water depth, optimal bottom type (i.e., hard bottom), good water quality (e.g., constant water flow, good light penetration), and limited biological stressors (e.g., coral predators and benthic space competitors such as algae, sponges gorgonians, and fire corals).
5. Coral relocations will be conducted the same day as their removal to the extent possible.
6. Corals to be transplanted will be held in separate containers (e.g., plastic buckets) to prevent colonies from contacting each other, kept submerged in water, and held in protected conditions (e.g., temporarily staged underwater in open or vented containers near the removal site for quick re-attachment following item removal or in a cooler or in shaded conditions on the support boat).

7. Before transplanting, all fouling organisms and sediment will be cleared from the substrate using wire brushes or scrapers.
8. Materials used to secure corals will consider the coral species, size of the coral transplant, substrate characteristics, and typical current or wave energy in the area.
9. Individual coral colonies or colony clusters that are transplanted will be field marked and GPS coordinates of their locations will be recorded in order to allow for future monitoring.
10. Success monitoring may be conducted using an ROV or by a scientific diver during subsequent munitions removal activities. Inspections will include, to the extent possible, documentation (including photos) of colony size and condition such as healthy and growing, partial or complete mortality, presence of disease, significant damage from coral predators (corallivores) such as fish, snails, or other invertebrates, and overgrowth or encrustation by organisms such as algae, sponges, tunicates, and cnidarians.
11. Any void created on the seafloor by an inadvertent impact during removal activities in seagrass will be backfilled with adjacent sediment so the grade of the impacted area is approximately flush with the surrounding grade.
12. Displaced rhizome segments or small seagrass plugs disturbed during removal activities will be re-planted by hand, using biodegradable pins, if necessary.
13. When larger subsurface items are investigated, an area of seagrass will be cut on three sides and rolled up. Once the anomaly has been investigated and potentially removed, the excavated area will be backfilled with the removed substrate and the seagrass rolled back into place and pinned (for plugs greater than approximately 8-in across) with biodegradable stakes.

<u>PDCs applicable to the use of divers</u>:

1. "Best diving practices" will be followed including the following for all activities requiring the use of divers:
    a. The dive team lead will make sure that underwater conditions (e.g., visibility, currents) and weather are suitable for diving to ensure diver safety and to avoid damaging ESA-listed corals or critical habitat.
    b. The point of water entry and exit will be carefully selected to avoid damaging coral
    c. Divers will make sure that all equipment is well secured before entering the water
    d. Divers will make sure that they are neutrally buoyant to the extent practical. If neutral buoyancy is not possible, divers will ensure their points of contact with the bottom or hard substrate are not on ESA-listed corals.
    e. Good finning practice and body control will be followed to avoid accidental contact with coral or stirring up the sediment.

f. Divers will limit physical contact with the benthic environment to the minimum extent needed to effectively conduct the work. As standard practice, impacts to any hard or soft corals shall be avoided to the greatest extent practicable.

g. Turbidity (from sediment resuspension) will be minimized to the extent possible during all underwater work activities. Although excessive turbidity is not expected to be generated by the underwater work activities, turbidity will be visually monitored and prudent measures will be taken to minimize turbidity generation to the extent possible.

3.3.2 Project-Specific Review and Step-Down Consultation Procedures

This programmatic consultation is based on the information available at the time of consultation. Later activities may include the need for BIPs, the use of fishing gear other than cast nets and fish traps for organism collection, the use of new technologies for survey or removal activities, or other activities within the scope of the proposed action for which we do not have detailed information at this time. Therefore, an activity-specific review must be completed to ensure all of the relevant PDCs are met and determine whether additional PDCs are required for a particular activity in UXO 16.

A project-specific review and step-down consultation request as specific projects or activities are planned for implementation must be submitted to NMFS ESA Interagency Cooperation Division, as appropriate. The Navy will certify compliance with the applicable PDCs along with the information described below to NMFS OPR in writing and send a copy to the Southeast Regional Office via email (nmfs.ser.esa.consultations@noaa.gov). The subject line should include a reference to "OPR-2018-00026, Programmatic Consultation with the Navy for Underwater Activities in UXO 16." The submission will include the following information:

1. Date sent to NMFS: This is the date the email was provided to NMFS.
2. Location: This should include the location where the activities will take place within UXO 16.
3. Transit routes: This should include information as to whether the transit routes to be used during a particular project and associated activities will be the same or different from the general transit routes analyzed in this Opinion. This information will enable NMFS to determine whether there may be changes to the action area that will affect the activity-specific effects analysis and the PDCs and thus determine if reinitiation of consultation is necessary.
4. PDCs met: Answer yes or no as to whether or not all of the applicable PDCs in this document will be met by the proposed project and associated activities within UXO 16 for the activities identified as not requiring further analysis.
5. Project-specific information should also be provided, including details of the activities to be conducted as part of the project and any proposed changes to the activities that were

analyzed in this Opinion or any new activities that will be associated with a particular project. This information will enable NMFS to determine the potential effects specific to a particular project on ESA resources in the action area and assess the risk to these resources because of the implementation of the project in UXO 16. The information will also enable NMFS to determine whether additional protective measures for avoidance and minimization of effects of a particular new activity or technology are required as part of a step-down consultation.

NMFS anticipates that step-down consultations may be required for the following activities either because of the uncertainty in estimating the extent of take of ESA-listed species as a result of the activity, the potential need for MMPA authorization for some activities such as the use of BIPs as a removal method, because of the potential for changes in some of these activities as technology evolves, or because details of the activity are not known at this time:

- BIPs
- Encapsulation
- Removal of in-water items known or suspected to present a significant explosive hazard
- The use of nets other than cast nets to collect organisms
- Tissue sampling from ESA-listed corals
- New temporary or permanent boat access points or improvements to four existing boat access points
- Seagrass and/or coral transplantation associated with the installation of new in-water structures
- The use of technology not described in the proposed action section for surveying and/or removal activities

Additionally, as noted above, this Opinion requires that the Navy make project-specific findings for every activity they carry out, review, permit or otherwise authorize to determine consistency with this Opinion, including its effects analyses. These reviews will determine the need for step-down consultation on an activity-specific basis. These reviews will also be compiled for annual review of this programmatic consultation.

3.3.3 Programmatic Review

The Navy and NMFS will conduct an annual programmatic review of the UXO 16 activities considered in this Opinion beginning one year after the issuance of this Opinion. This review will evaluate, among other things, whether the scope of the operations being implemented is consistent with the description of the proposed activities; whether the nature and scale of effects predicted continue to be valid; whether the PDCs are being complied with and continue to be appropriate; and whether the project-specific and step-down consultation procedures are being complied with and are effective.

To assist in this annual review, the Navy will submit a summary review 30 days prior to the end of the first 12-month period after the issuance of the Opinion, and 30 days prior to the close of all subsequent 12-month periods. The Navy will submit a summary of the in-water activities conducted during each 12-month period in UXO 16; information regarding the PDCs implemented for each activity and their efficacy in avoiding and minimizing impacts of the program on ESA-listed species and their designated critical habitat; any issues identified by the dedicated observer, vessel captain or other crew member, divers or other personnel engaged in the activity in implementing avoidance and minimization measures; copies of sighting logs for marine mammals and sea turtles; and monitoring and reporting of take of ESA-listed species included in an ITS. There may be more or less reporting requirements as the program proceeds.

4 ACTION AREA

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 C.F.R. §402.02).

The Navy identified marine access points for vessels that are towed in and out of the water by terrestrial vehicles during site operations within UXO 16 (see Section 3.2.5). Other marine access points around Vieques that are outside UXO 16 include the Esperanza Pier boat ramp and the Isabella boat ramp.

In the past, vessels used for marine operations have also transited between mainland Puerto Rico, particularly Fajardo, and Vieques. Therefore, we include possible transit routes between Fajardo and Vieques in the action area as well. This includes the former Roosevelt Roads Naval Station (identified in Figure 2 as U.S. Naval Activity Puerto Rico).

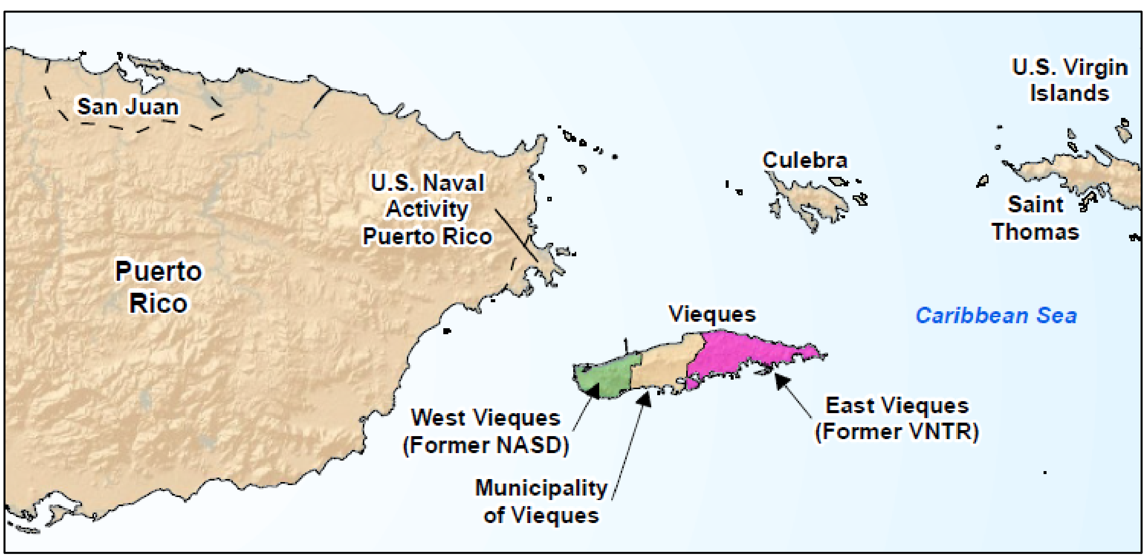


Figure 2. Map showing the location of Vieques in relation to mainland Puerto Rico (adapted from CH2M Hill 2018)

5 POTENTIAL STRESSORS

Stressors are any physical, chemical, or biological agent, environmental condition, external stimulus or event that may induce an adverse response in either an ESA-listed species or its designated critical habitat (Schulte 2014). The action consists of location and removal of underwater surface and subsurface MEC/MPPEH, collection of samples (water, sediment, and biota), installation and maintenance of in-water structures, underwater investigations to look for MEC, boating operations associated with the other activities, and transplant of coral and seagrass associated with some removal activities. The major categories of stressors from these activities (Table 1) are:

- strikes (e.g., vessels, ROVs, towed equipment)
- vessel anchoring, propeller wash and scarring, and grounding
- vessel discharges and marine debris
- sound from different sources (e.g., vessel noise, echosounders and other vessel navigational equipment, electromagnetic and other sensors used during underwater investigations, nonintentional detonation)
- entanglement and entrapment (e.g., in gear used to collect biotic samples, in tackle associated with in-water structures such as buoys, in waterway barriers, with towlines and cables of ROVs and towed sensors/equipment)
- sediment resuspension and transport from various activities (e.g., propeller wash, sediment sampling, anchor installation for in-water structures, use of bottom-operated sensor equipment)
- habitat loss and/or damage (e.g., in-water structure installation, bottom moving sensor equipment, use of lift bags/balloons or tripods for MEC removal, nonintentional detonation, temporary marker placement during investigation and removal activities, diver breakage and abrasion)
- bycatch from the use of nets and traps to sample fish and invertebrates and organism collection and transplant (e.g., coral tissue sampling, fish collection, removal with MEC/MPPEH, transplanting coral and seagrass, nonintentional detonation)
- leaching of metals from sacrificial anodes (e.g., boat motors, in-water structures) and contaminants released from MEC/MPPEH during removal activities and associated sediment sampling.

Table 1. Summary of Stressors Associated with the Categories of Activities Proposed

	Activity Category					
Stressor	Vessel Operation*	Diver Operation**	Location and Removal	Sample Collection	In-Water Structures	Underwater Investigation
Strikes/ Collisions	X		X			X
Vessel Anchoring/Propeller Wash/Scarring/ Grounding	X					
Vessel Discharges/ Marine Debris	X					
Noise	X		X	X	X	X
Entanglement/ Entrapment			X	X	X	X
Sediment Resuspension	X	X	X	X	X	X
Habitat Loss or Damage	X	X	X	X	X	X
Bycatch				X		
Organism Collection and Transplant			X	X	X	
Metal Leachate/Contaminant Release	X		X	X	X	

* Vessel Operation and associated stressors apply across all activities

** Diver Operation and associated stressors apply to location and removal, sample collection, in-water structures, and underwater investigation activities.

6 Endangered Species Act Resources That May Be Affected

This section identifies the ESA-listed species and designated critical habitat that potentially occur within the action area (Table 2) that may be affected by the proposed underwater activities in UXO 16. This section first identifies the species and designated critical habitat that may be affected, but are not likely to be adversely affected by the proposed action. The remaining species and designated critical habitat deemed likely to be adversely affected by one or more of the proposed activities in the action area considered in this Opinion are carried forward through the remainder of this Opinion.

Table 2. Threatened and Endangered Species That May Be Affected by the Proposed Action

Species	**ESA Status**	Recovery Plan	Critical Habitat
	Marine Mammals		
Blue whale (*Balaenoptera musculus*)	E – 35 FR 18319, December 2,1970	07/1998	----
Fin whale (*Balaenoptera physalus*)	E – 35 FR 18319, December 2,1970	75 FR 47538	----
Sei whale (*Balaenoptera borealis*)	E – 35 FR 18319, December 2,1970	76 FR 43985	----
Sperm whale (*Physeter microcephalus*)	E – 35 FR 18319, December 2,1970	75 FR 81584	----
	Fish		
Nassau grouper (*Epinephelus striatus*)	T – 81 FR 42268, June 29, 2016	----	----
Giant manta ray (*Manta birostris*), Southwest Distinct Population Segment (DPS)	T – 83 FR 2916, January 22, 2018	----	----
Scalloped hammerhead shark (*Sphyrna lewini*), Central and Southwest Atlantic Distinct Population Segment (DPS)	T – 79 FR 38214, July 3, 2014	----	----
Oceanic whitetip shark (*Carcharinus longimanus*)	T – 83 FR 4153, January 30, 2018	----	----
	Sea Turtles		
Green sea turtle (*Chelonia mydas*), North Atlantic DPS	T – 81 FR 20057, April 6, 2016 (original listing 1978)	63 FR 28359	Not in action area
Green sea turtle (*Chelonia mydas*), South Atlantic DPS	T – 81 FR 20057, April 6, 2016	63 FR 28359	----
Hawksbill sea turtle (*Eretmochelys imbricata*)	E – 35 FR 8491, June 2, 1970	12/1993	Not in action area
Leatherback sea turtle (*Dermochelys coriacea*)	E – 35 FR 8491, June 2, 1970	63 FR 28359	Not in action area
Loggerhead sea turtle (*Caretta caretta*), Northwest Atlantic Ocean DPS	T – 76 FR 58868, September 22, 2011 (original listing 1978)	63 FR 28359	Not in action area

Species	ESA Status	Recovery Plan	Critical Habitat
Corals			
Elkhorn coral (*Acropora palmata*)	T – 71 FR 26852, May 9, 2006, and 79 FR 53852, September 10, 2014	80 FR 12146	73 FR 72210 (Puerto Rico unit)
Staghorn coral (*Acropora cervicornis*)	T – 71 FR 26852, May 9, 2006, and 79 FR 53852, September 10, 2014	80 FR 12146	73 FR 72210 (Puerto Rico unit)
Lobed star coral (*Orbicella annularis*)	T – 79 FR 53852, September 10, 2014	----	----
Boulder star coral (*Orbicella franksi*)	T – 79 FR 53852, September 10, 2014	----	----
Mountainous star coral (*Orbicella faveolata*)	T – 79 FR 53852, September 10, 2014	----	----
Pillar coral (*Dendrogyra cylindrus*)	T – 79 FR 53852, September 10, 2014	----	----
Rough cactus coral (*Mycetophyllia ferox*)	T – 79 FR 53852, September 10, 2014	----	----
T = threatened, E = endangered			

Marine mammals protected under the Marine Mammal Protection Act (MMPA) including the ESA-listed whales identified in Table 2, and species such as bottlenose dolphins (*Tursiops truncatus*) occur in the action area. If these or other non-ESA-listed marine mammals may be adversely affected by the proposed action, a take authorization under the MMPA may be necessary. OPR's Permits and Conservation Division should be contacted for more information regarding MMPA requirements at 301-427-8401 (see also https://www.fisheries.noaa.gov/topic/marine-mammal-protection). If MMPA authorization is required for any of the activities that will be conducted by the Navy in UXO 16, an ESA section 7 consultation would also be required for the issuance of an MMPA permit, authorization, or rule-making by the Permits and Conservation Division.

6.1 Species and Critical Habitat Not Likely to be Adversely Affected

NMFS uses two criteria to identify the ESA-listed or designated critical habitat that are not likely to be adversely affected by the action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that co-occurs with a stressor of the action but is not likely to respond to the stressor is also not likely to be adversely affected by the action. We applied these criteria to the ESA-listed species in Table 2 and we summarize our results below.

In the case of the proposed action, ESA-listed species and designated critical habitat occur in waters affected by the underwater activities detailed in Section 3.2 that will take place in UXO 16.

The probability of an effect on a species or designated critical habitat is a function of exposure intensity and susceptibility of a species to a stressor's effects (i.e., probability of response). An action warrants a "may affect, not likely to adversely affect" finding when its effects are wholly *beneficial, insignificant*, or *discountable*.

Beneficial effects have an immediate positive effect without any adverse effects to the species or habitat. *Insignificant* effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen, but will not rise to the level of constituting an adverse effect. *Discountable* effects are those that are extremely unlikely to occur. For an effect to be discountable, there must be a plausible adverse effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did affect a listed species), but it is very unlikely to occur (NMFS and USFWS 1998).

6.1.1 Fin, Sei, and Blue Whales

Fin, sei, and blue whales are offshore, deep-water species. Fin and sei whales have only been observed in Puerto Rico north of Mona Island and south of Cayo Ratones, Salinas, and records indicate blue whales are not regular inhabitants of the Caribbean (GMI 2003 cited in; CH2M Hill 2018). The Navy does not have data indicating any of these species have been observed during in-water activities associated with underwater surveys, including the WAA, and cleanup activities in UXO 16. A review of our consultation files, particularly those with the Puerto Rico Aqueduct and Sewer Authority that include monitoring of offshore sewage outfalls on a regular basis, indicate that these three species are not reported in waters off Puerto Rico. Humpback whales and other non-ESA-listed species are commonly reported and sperm whales are observed infrequently.

The majority of activities that are part of this consultation will be conducted in nearshore, shallow waters of UXO 16 and are not expected to have any effect on these three whale species. Removal, survey, and sampling activities at the three offshore anchorage areas are the only activities that will occur in deeper waters. Vessel transit to and from these offshore areas, as well as between ports and harbors in the action area that includes the main island of Puerto Rico, could result in encounters with ESA-listed whale species. However, the rarity of these three species and the fact that reported sightings do not include any areas that fall within the action area for this consultation mean that vessel strikes or other effects to fin, sei, and blue whales as a result of the proposed action are extremely unlikely to occur and therefore discountable. Therefore, we believe the proposed action is not likely to adversely affect these three species of ESA-listed whales.

6.1.2 Loggerhead Sea Turtles

There are no reports of nesting of loggerhead sea turtles on the beaches of Vieques based on information from the Navy associated with the sea turtle conservation project in partnership with USFWS and PRDNER to monitor beaches around the island for ten months of the year. Limited loggerhead nesting has been reported on the east coast of mainland Puerto Rico and on Culebra Island, but is apparently not frequent. Loggerhead sea turtles could be present in nearshore and offshore waters where proposed removal, survey, and sampling activities in UXO 16. Loggerhead hatchlings use floating mats of *Sargassum* while adults and juveniles may be present along the shelf edge and in shallow habitats such as estuaries, reefs, and natural and artificial hard bottom. However, stranding data from the PRDNER indicate that no loggerhead sea turtles have been reported as stranded from 1989-2009, indicating that the species is not likely to be found in UXO 16 during activities conducted as part of the proposed action.

Stressors from vessel operation and associated discharges and potential generation of marine debris, noise, entanglement and entrapment, bycatch in fishing gear, and sediment resuspension and transport during removal, survey, and sampling activities have the potential to affect juvenile and adult life stages of loggerhead sea turtles. Vessel transit to, from, and within UXO 16, as well as between ports and harbors in the action area, could result in encounters with loggerhead sea turtles. Stranding and nesting data from PRDNER indicate that this species can occasionally be found along the eastern coast of the main island of Puerto Rico, including nesting on some beaches, but nesting and stranding events involving the species do not occur frequently. Therefore, because of the rarity of loggerhead sea turtles around Puerto Rico and the lack of nesting, stranding and sighting data indicating they are present in the action area for this consultation mean that vessel strikes or other effects as a result of the proposed action are extremely unlikely to occur and therefore discountable. Thus, we believe the proposed action is not likely to adversely affect loggerhead sea turtles.

6.1.3 ESA-Listed Elasmobranchs

Giant manta rays are typically found offshore in the open ocean though these animals are sometimes found around nearshore reefs and estuarine waters, which are some of the habitats present in the action area. Giant manta rays feed in the water column on plankton. Giant manta ray have been observed infrequently near the entrance to San Juan Bay particularly near channel marker buoys by NMFS biologists and infrequent observations of this species have also been reported in deeper waters off bays and over deep reefs around the U.S. Virgin Islands (USVI; A. Dempsey, BioImpact, personal communications to L. Carrubba, NMFS, January 26, 2018, and February 26, 2018; R. Nemeth, University of the Virgin islands, personal communication to L. Carrubba, NMFS, January 26, 2018). Because the action area has similar habitat as the sites around the USVI where these animals have occasionally been sighted, it is possible that they transit through the action area periodically. However, the Navy and its contractors have not documented sightings of giant manta rays during numerous in-water surveys conducted as part of the on-going evaluation of potential MEC/MPPEH.

The oceanic whitetip shark is usually found offshore in the open ocean, along the continental shelf, or around oceanic islands in waters from the surface to at least 152 m in depth. Oceanic whitetip sharks are highly mobile and prefer open ocean conditions, including for foraging. Shark tagging data show movements by juveniles of this species in the Gulf of Mexico, along the east coast of Florida, Mid-Atlantic Bight, Cuba, Lesser Antilles, central Caribbean Sea, from east to west along the equatorial Atlantic, and off Brazil, Haiti, and Bahamas (Young et al. 2017). Fisheries data also indicate that, while catch of this species has declined, it has been part of fishery landings in the U.S. Caribbean (Young et al. 2017) meaning that the species is likely to be present in offshore waters of Puerto Rico. As for giant manta rays, the Navy and its contractors have not reported this species during in-water work within UXO 16.

Data from the Marine Recreational Information Program (MRIP) from Puerto Rico from 2001 – 2016 show 797 scalloped hammerhead sharks were landed by recreational charter boats using vertical line gear within Puerto Rico's territorial waters, which extend to 9 nm from shore. The greatest number of scalloped hammerhead sharks, 516, were captured in 2003. The other landings were from 2004 (44), 2006 (30), 2012 (98), and 2016 (109). Landed sharks ranged in length from 600 – 800 millimeters (mm), meaning they were likely neonates or juveniles as maturity is reached when males are approximately 1,219 mm and females are 1,981 mm. At least some of the sharks may have been misidentified and were actually bonnetheads. Others were included in a general hammerhead shark category and could be species other than scalloped hammerhead (M. Wunderlich, NMFS SERO, pers. comm. to L. Carrubba, NMFS OPR, October 13, 2017). Adult sharks tend to be more common in offshore waters while neonates and juveniles are more common in nearshore waters in areas where they occur. There are limited data from the U.S. Caribbean indicating that two bays, one in St. Thomas and one in St. John, USVI serve as nursery habitat for neonate scalloped hammerhead sharks (DeAngelis 2006). There are no data indicating there are nursery areas in bays around Puerto Rico and the species has not been reported during in-water work conducted by the Navy and its contractors in UXO 16 to date.

Stressors from vessel operation and associated discharges and potential generation of marine debris, noise, and entanglement are those with the potential to affect giant manta ray, oceanic whitetip sharks, and scalloped hammerhead sharks. In addition, because juveniles and neonates of scalloped hammerhead sharks have the potential to be present in bays, bycatch during sampling using fishing gear, as well as sediment resuspension and transport, and other activities resulting in habitat loss or degradation also have the potential to affect these life stages of the species. Vessel transit to, from, and within UXO 16, as well as between ports and harbors in the action area, could result in encounters with giant manta rays, oceanic whitetip sharks, and scalloped hammerhead sharks. However, because of the apparent rarity of these species in the action area and the lack of sightings reports or other data indicating they are present in UXO 16, vessel strikes or other effects to these species as a result of the action are extremely unlikely to occur and therefore discountable. Therefore, we believe the proposed action is not likely to adversely affect giant manta rays, oceanic whitetip sharks, and scalloped hammerhead sharks.

6.2 Status of Species and Critical Habitat Likely to be Adversely Affected

This Opinion examines the status of sperm whales; green (North and South Atlantic DPSs), leatherback, and hawksbill sea turtles; Nassau grouper; elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, and boulder star coral; and designated critical habitat for elkhorn and staghorn coral (Puerto Rico unit) that may be affected by the action.

The evaluation of adverse effects in this Opinion begins by summarizing the biology and ecology of those species that are likely to be adversely affected and what is known about their life histories in the action area and the condition of designated critical habitat within the applicable critical habitat unit and in the action area. The status is determined by the level of risk that the ESA-listed species and designated critical habitat face based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This helps to inform the description of the species' current "reproduction, numbers or distribution" that is part of the jeopardy determination as described in 50 C.F.R. §402.02. This section also examines the condition of critical habitat throughout the designated area (such as various watersheds and coastal and marine environments that make up the designated area), and discusses the condition and current function of designated or proposed critical habitat, including the essential physical and biological features that contribute to that conservation value of the critical habitat. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on the NMFS Web site: [https://www.fisheries.noaa.gov/topic/endangered-species-conservation].

6.2.1 Sperm Whale

The sperm whale is a widely distributed species found in all major oceans (Figure 3). 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, December 2, 1970].

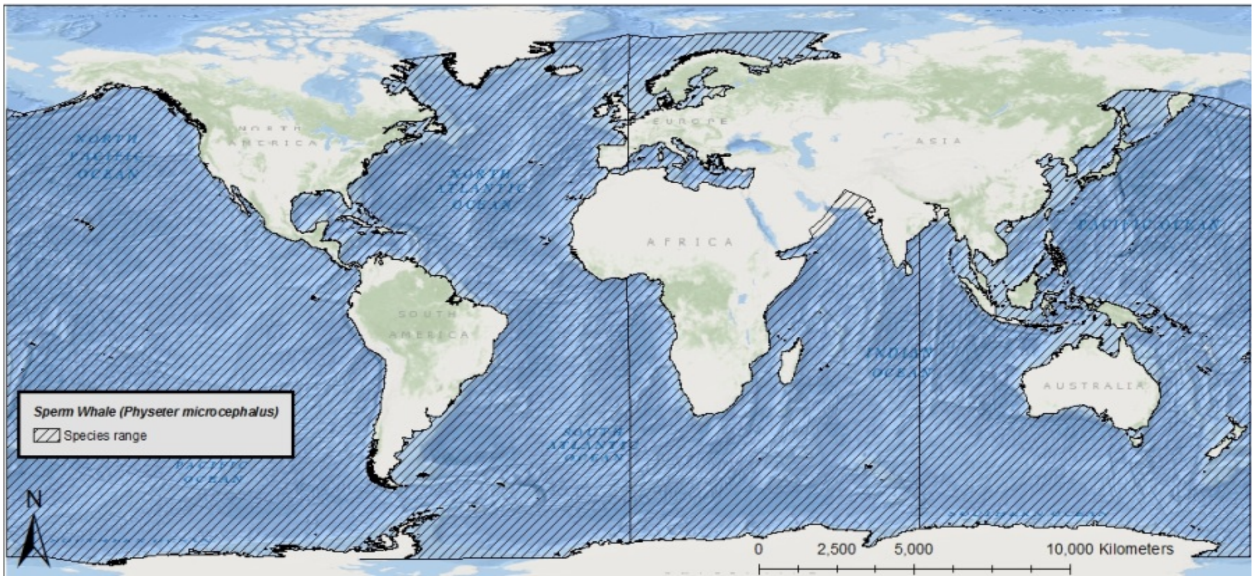


Figure 3. Map identifying the range of the endangered sperm whale

Life History and Population Dynamics

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, disperse more widely, and can mate with multiple female populations throughout a lifetime.

The average lifespan of sperm whales is estimated to be at least 50 years (Whitehead 2009). They have a gestation period of one to one and a half years, and calves nurse for approximately two years. Sexual maturity is reached between seven and thirteen years of age for females with an average calving interval of four to six years. Male sperm whales reach full sexual maturity in their twenties. Sperm whales have a strong preference for waters deeper than 1,000 m (Reeves and Whitehead 1997; Watkins 1977), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are occasionally found in waters less than 300 m in depth (Rice 1989; Clarke 1956). Sperm whales have been observed near Long Island, New York, in water between 40-55 m deep (Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956). Such areas include oceanic islands and along the outer continental shelf. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed primarily on squid; other prey includes octopus and demersal fish (including teleosts and elasmobranchs).

The sperm whale is the most abundant of the large whale species, with total abundance estimates between 200,000 and 1,500,000. The most recent estimate indicated a global population of between 300,000 and 450,000 individuals (Whitehead 2009). The higher estimates may be approaching population sizes prior to commercial whaling, the reason for ESA listing. However, there is insufficient data to evaluate trends in abundance and growth rates of sperm whales at this time.

There are six recognized stocks of sperm whales that exist in U.S. waters: California/Oregon/Washington (N=2,106, N_{min}=1,332), Hawaii (N=3,354; N_{min}=2,539), Northern Gulf of Mexico (N=763, N_{min}=560), North Pacific (no reliable population estimate at this time), North Atlantic (N=2,288 (underestimate); N_{min}=1,815), and Puerto Rico and the U.S. Virgin Islands (insufficient population data).

Ocean-wide genetic studies indicate sperm whales have low genetic diversity, suggesting a recent bottleneck, but strong differentiation between matrilineally related groups (Lyrholm and Gyllensten 1998). Consistent with this, two studies of sperm whales in the Pacific indicate low genetic diversity (Mesnick et al. 2011; Rendell et al. 2012). Furthermore, sperm whales from the Gulf of Mexico, the western North Atlantic, the North Sea, and the Mediterranean Sea all have been shown to have low levels of genetic diversity (Engelhaupt et al. 2009). As none of the stocks for which data are available have high levels of genetic diversity, the species may be at some risk to inbreeding and 'Allee' effects, although the extent to which is currently unknown.

Sperm whales have a global distribution and can be found in relatively deep waters in all ocean basins (Figure 3). While both males and females can be found in latitudes less than 40º, only adult males venture into the higher latitudes near the poles.

In the western North Atlantic, sperm whales range from Greenland south into the Gulf of Mexico and the Caribbean, where they are common, especially in deep basins off of the continental shelf (Romero et al. 2001; Wardle et al. 2001). The northern distributional limit of female/immature pods is probably around Georges Bank or the Nova Scotian shelf (Whitehead et al. 1991). Seasonal aerial surveys confirm that sperm whales are present in the northern Gulf of Mexico in all seasons (Mullin et al. 1994; Hansen et al. 1996). Sperm whales distribution follows a distinct seasonal cycle, concentrating east-northeast of Cape Hatteras in winter and shifting northward in spring when whales are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-Atlantic Bight. In the eastern Atlantic, mature male sperm whales have been recorded as far north as Spitsbergen (Øien 1990). Recent observations of sperm whales and stranding events involving sperm whales from the eastern North Atlantic suggest that solitary and paired mature males predominantly occur in waters off Iceland, the Faroe Islands, and the Norwegian Sea (Øien 1990; Gunnlaugsson and Sigurjónsson 1990; Christensen et al. 1992b; Christensen et al. 1992a).

Vocalization and Hearing

Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broadband clicks in the frequency range of 10 hertz (Hz) to 30 kilohertz (kHz) that can be extremely loud for a biological source (André et al. 2017). Evidence suggests that the clicks produced during foraging dives are directional with an intense, forward-directed beam at levels as high as 236 decibels (dB) re: 1 micro Pascal (μPa) at 1 m (Mohl et al. 2003). Most of the energy in sperm whale clicks is concentrated at around 2-4 kHz and 10-16 kHz (Goold and Jones 1995; Weilgart and Whitehead 1993; NMFS 2006d). The multipulsed nature of sperm whale clicks led to the dominating theory of sound production mechanics by Norris and Harvey (1972), who explained the interpulse interval of the click by properties of the nasal anatomy (Mohl et al. 2003). This theory has been supported by sound-transmission experiments within the spermaceti complex (Mohl et al. 2003). Clicks are also used in short patterns (codas) during social behavior and intragroup interactions (Weilgart and Whitehead 1993) and may also aid in intra-specific communication. Another class of sound, “squeals”, are produced with frequencies of 100 Hz to 20 kHz (e.g., Weir et al. 2007).

Our understanding of sperm whale hearing stems largely from the sounds they produce. The only direct measurement of hearing was from a young stranded individual from which auditory evoked potentials were recorded (Carder and Ridgway 1990). From this whale, responses support a hearing range of 2.5-60 kHz. However, behavioral responses of adult, free-ranging individuals also provide insight into hearing range; sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins et al. 1985; 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). Because they spend large amounts of time at depth and use low-frequency sound, sperm whales are likely to be susceptible to low frequency sound in the ocean (Croll et al. 1999).

Status

The sperm whale is endangered because of past commercial whaling. Although the aggregate abundance worldwide is probably at least several hundred thousand individuals, the extent of depletion and degree of recovery of populations are uncertain. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead 2003). Continued threats to sperm whale populations include ship strikes, entanglement in fishing gear, competition for resources due to overfishing, pollution, loss of prey and habitat due to climate change, and noise. The species’ large population size shows that it is somewhat resilient to current threats.

190,000 sperm whales were estimated to have been in the entire North Atlantic, but CPUE data from which this estimate is derived are unreliable according to the IWC (Perry et al. 1999). The total number of sperm whales in the western North Atlantic is unknown (Waring et al. 2008). The best available current abundance estimate for western North Atlantic sperm whales is 4,804 based on 2004 data. The best available estimate for Northern Gulf of Mexico sperm whales is

1,665, based on 2003-2004 data, which are insufficient data to determine population trends (Waring et al. 2008). Sperm whale were widely harvested from the northeastern Caribbean (Romero et al. 2001) and the Gulf of Mexico where sperm whale fisheries operated during the late 1700s to the early 1900s (Townsend 1935; NMFS 2006).

Critical Habitat

No critical habitat has been designated for the sperm whale.

Recovery Goals

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. See the 2010 Final Recovery Plan for the sperm whale for complete down listing/delisting criteria for both of the following recovery goals.

1. Achieve sufficient and viable populations in all ocean basins.
2. Ensure significant threats are addressed.

6.2.2 General Threats Faced by Green (North and South Atlantic DPS) and Hawksbill Sea Turtles

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

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

Non-Fishery In-Water Activities

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

Coastal Development and Erosion Control

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

Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., dichlorodiphenyltrichloroethane [DDT], polychlorinated biphenyls [PCB], and perfluorinated chemicals [PFC]), and others that may cause adverse health effects to sea turtles (Iwata et al. 1993; Grant and Ross 2002; Garrett 2004; Hartwell 2004). 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 1997). Hydrocarbons also have the potential to affect prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area.

The April 20, 2010, explosion of the DEEPWATER HORIZON (DWH) oilrig affected sea turtles in the Gulf of Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (DWH Trustees 2015). Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. The spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will affect 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. Marine debris is a problem due primarily to sea turtles ingesting debris and blocking the digestive tract, causing death or serious injury (Lutcavage et al. 1997; Laist et al. 1999). Schuyler et al. (2015) estimated that, globally, 52 percent of individual sea turtles have ingested marine debris. Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives; this figure is supported by data from Lazar and Gračan (2011), who found 35 percent of loggerheads had plastic in their gut. A Brazilian study found that 60 percent of stranded green sea turtles had ingested marine debris (Bugoni et al. 2001). Loggerhead sea turtles had a lesser frequency of marine debris ingestion. Plastic may be ingested out of curiosity or due to confusion with prey items. Marine debris consumption has been shown to depress growth rates in post-hatchling loggerhead sea turtles, increasing the time required to reach sexual maturity and increasing predation risk (McCauley and Bjorndal 1999). Sea turtles can also become entangled and die in marine debris, such as discarded nets and monofilament line (NRC 1990; Lutcavage et al. 1997; Laist et al. 1999).

Climate Change

See Section 7.2.1 for a discussion of the threat of climate change to 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. These mammals, as well as ghost crabs, laughing gulls, and the exotic South American fire ant (*Solenopsis invicta*), prey upon emergent hatchlings. In addition to natural predation, direct harvest of eggs and adults from beaches in foreign countries continues to be a problem for various sea turtle species throughout their ranges (NMFS and USFWS 2008).

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

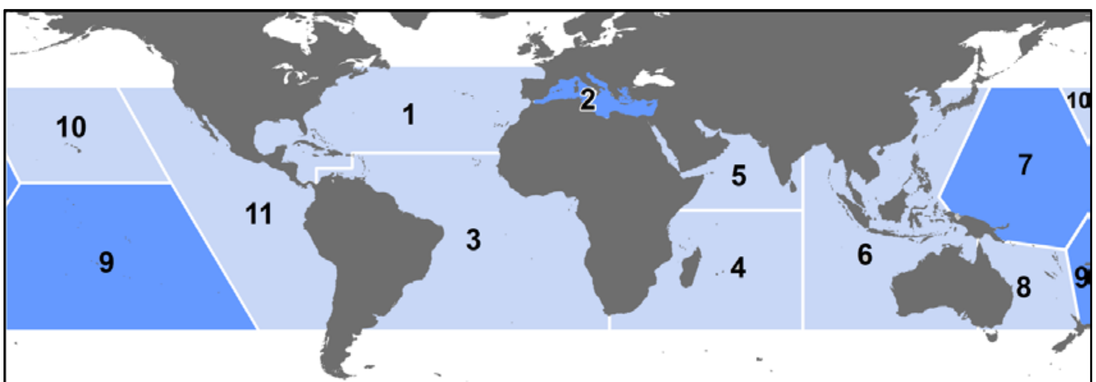
6.2.2.1 Status of Green Sea Turtle (North and South Atlantic DPSs)

The species was listed under the ESA on July 28, 1978 (43 FR 32800). The species was separated into two listing designations: endangered for breeding populations in Florida and the Pacific coast of Mexico and threatened in all other areas throughout its range. On April 6, 2016, NMFS listed 11 DPSs of green sea turtles as threatened or endangered under the ESA (Figure 4; 81 FR 20057).

Species Description and Life History

The green sea turtle (*Chelonia mydas*) is the largest of the hardshell marine turtles. It has a circumglobal distribution, occurring throughout nearshore tropical, subtropical and, to a lesser extent, temperate waters.

Eight DPSs are listed as threatened: Central North Pacific, East Indian-West Pacific, East Pacific, North Atlantic, North Indian, South Atlantic, Southwest Indian, and Southwest Pacific. Three DPSs are listed as endangered: Central South Pacific, Central West Pacific, and Mediterranean.



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

Figure 4. Map depicting DPS boundaries for green turtles.

Age at first reproduction for females is 20 - 40 years. Green sea turtles lay an average of three nests per season with an average of 100 eggs per nest. The remigration interval (i.e., return to natal beaches) is 2 - 5 years. Nesting occurs primarily on beaches with intact dune structure, native vegetation and appropriate incubation temperatures during summer months. 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. Adult turtles exhibit site fidelity and migrate hundreds to thousands of kilometers from nesting beaches to foraging areas. Green sea turtles spend the majority of their lives in coastal foraging grounds, which include open coastlines and protected bays and lagoons. Adult green turtles feed primarily on seagrasses and algae, although they also eat jellyfish, sponges and other invertebrate prey.

Population dynamics

Abundance

Worldwide, nesting data at 464 sites indicate that 563,826 to 564,464 females nest each year (Seminoff et al. 2015).

<u>North Atlantic DPS</u>

Compared to other DPSs, the North Atlantic DPS exhibits the highest nester abundance, with approximately 167,424 females at 73 nesting sites; Figure 5), and available data indicate an increasing trend in nesting. The largest nesting site in the North Atlantic DPS is in Tortuguero, Costa Rica, which hosts 79 percent of nesting females for the DPS (Seminoff et al. 2015).

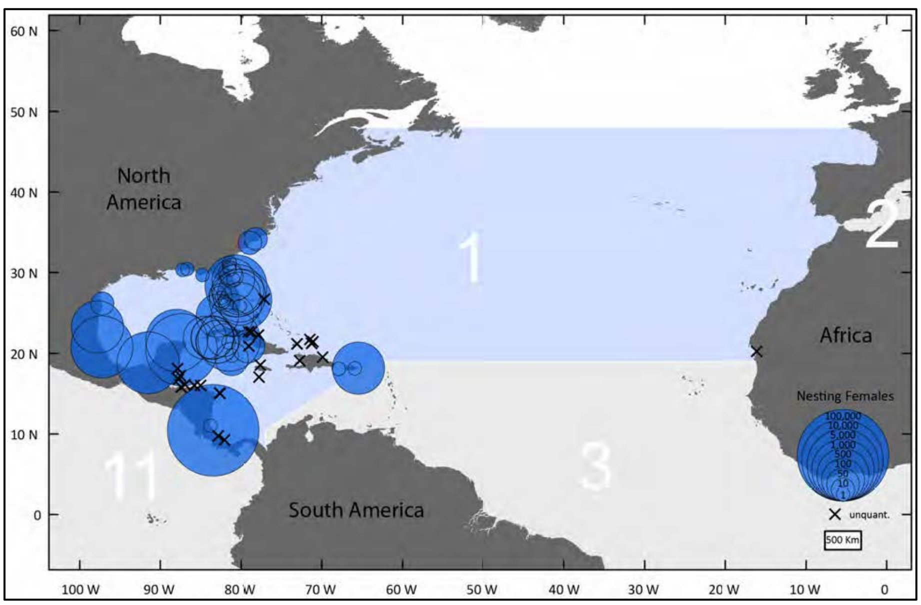


Figure 5. Geographic range of the North Atlantic DPS, with location and abundance of nesting females (from Seminoff et al. 2015)

South Atlantic DPS

The South Atlantic DPS has 51 nesting sites, with an estimated nester abundance of 63,332. The largest nesting site is at Poilão, Guinea-Bissau, which hosts 46 percent of nesting females for the DPS (Seminoff et al. 2015).

Population Growth Rate

North Atlantic DPS

For the North Atlantic DPS, the available data indicate an increasing trend in nesting. There are no reliable estimates of population growth rate for the DPS as a whole, but estimates have been developed at a localized level. Modeling by Chaloupka et al. (2008) using data sets of 25 years or more show the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent.

South Atlantic DPS

There are 51 nesting sites for the South Atlantic DPS, and many have insufficient data to determine population growth rates or trends. Of the nesting sites where data are available, such as Ascension Island, Suriname, Brazil, Venezuela, Equatorial Guinea, and Guinea-Bissau, there is evidence that population abundance is increasing.

Genetic Diversity

Globally, the green turtle is divided into eleven distinct population segments; available information on the genetic diversity for the North Atlantic and South Atlantic distinct population segments is presented below.

North Atlantic DPS

The North Atlantic DPS has a globally unique haplotype, which was a factor in defining the discreteness of the population for the DPS. Evidence from mitochondrial DNA studies indicates that there are at least four independent nesting subpopulations in Florida, Cuba, Mexico and Costa Rica (Seminoff et al. 2015). More recent genetic analysis indicates that designating a new western Gulf of Mexico management unit might be appropriate (Shamblin et al. 2015).

South Atlantic DPS

Individuals from nesting sites in Brazil, Ascension Island, and western Africa have a shared haplotype found in high frequencies. Green turtles from rookeries in the eastern Caribbean however, are dominated by a different haplotype.

Distribution

North Atlantic DPS

Green turtles from the North Atlantic DPS range from the boundary of South and Central America (7.5°N, 77°W) in the south, throughout the Caribbean, the Gulf of Mexico, and the U.S.

Atlantic coast to New Brunswick, Canada (48°N, 77°W) in the north. The range of the DPS then extends due east along latitudes 48°N and 19°N to the western coasts of Europe and Africa (Figure 11).

South Atlantic DPS

The range of the South Atlantic DPS begins at the border of Panama and Colombia at 7.5^{o}N, 77^{o}W, heads due north to 14^{o}N, 77^{o}W, then east to 14^{o}N, 65.1^{o}W, then north to 19^{o}N, 65.1^{o}W, and along 19^{o}N latitude to Mauritania in Africa. It extends along the coast of Africa to South Africa, with the southern border being 40^{o}S latitude (Figure 4).

Status

We used information available in the 2007 5-Year Review (NMFS and USFWS 2007) and 2015 Status Review (Seminoff et al. 2015) to summarize the status of the species, as follows.

Once abundant in tropical and subtropical waters, green sea turtles worldwide exist at a fraction of their historical abundance, as a result of over-exploitation. Globally, egg harvest, the harvest of females on nesting beaches and directed hunting of turtles in foraging areas remain the three greatest threats to their recovery. In addition, bycatch in drift net, long-line, set-net, pound-net and trawl fisheries kill thousands of green sea turtles annually. Increasing coastal development (including beach erosion and re-nourishment, construction and artificial lighting) threatens nesting success and hatchling survival. On a regional scale, the different DPSs experience these threats as well, to varying degrees. Differing levels of abundance combined with different intensities of threats and effectiveness of regional regulatory mechanisms make each DPS uniquely susceptible to future perturbations.

North Atlantic DPS

Historically, green turtles in the North Atlantic DPS were hunted for food, which was the principle cause of the population's decline. Apparent increases in nester abundance for the North Atlantic DPS in recent years are encouraging but must be viewed cautiously, as the datasets represent a fraction of a green sea turtle generation, up to 50 years. While the threats of pollution, habitat loss through coastal development, beachfront lighting, and fisheries bycatch continue, the North Atlantic DPS appears to be somewhat resilient to future perturbations.

South Atlantic DPS

Though there is some evidence that the South Atlantic DPS is increasing, there is a considerable amount of uncertainty over the impacts of threats to the South Atlantic DPS. The DPS is threatened by habitat degradation at nesting beaches, and mortality from fisheries bycatch remains a primary concern.

Critical Habitat

There is no designated critical habitat for the South Atlantic DPS. The North Atlantic DPS includes green sea turtle critical habitat designated on September 2, 1998, which includes waters surrounding Culebra Island, Puerto Rico, which is outside the action area of this consultation.

Recovery Goals

See the 1998 and 1991 recovery plans for the Pacific, East Pacific, and Atlantic populations of green turtles for complete down-listing/delisting criteria for recovery goals of the species. Broadly, recovery plan goals emphasize the need to protect and manage nesting and marine habitat, protect and manage populations on nesting beaches and in the marine environment, increase public education, and promote international cooperation on sea turtle conservation topics. For the Atlantic, which encompasses the North and South Atlantic DPSs, the recovery objectives are:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years. Nesting data must be based on standardized surveys.
- At least 25 percent (105 km) of all available nesting beaches (420 km) is in public ownership and encompasses at least 50 percent of the nesting activity.
- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.
- All priority one tasks have been successfully implemented.

6.2.2.2 Status of Leatherback Sea Turtles

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 sea turtle is unique among sea turtles for its large size, wide distribution (due to thermoregulatory systems and behavior), and lack of a hard, bony carapace. It ranges from tropical to subpolar latitudes, worldwide (Figure 6). Leatherbacks are the largest living turtle, reaching lengths of six feet long, and weighing up to one ton. Leatherback sea turtles have a distinct black leathery skin covering their carapace with pinkish white skin on their belly (Figure 6).

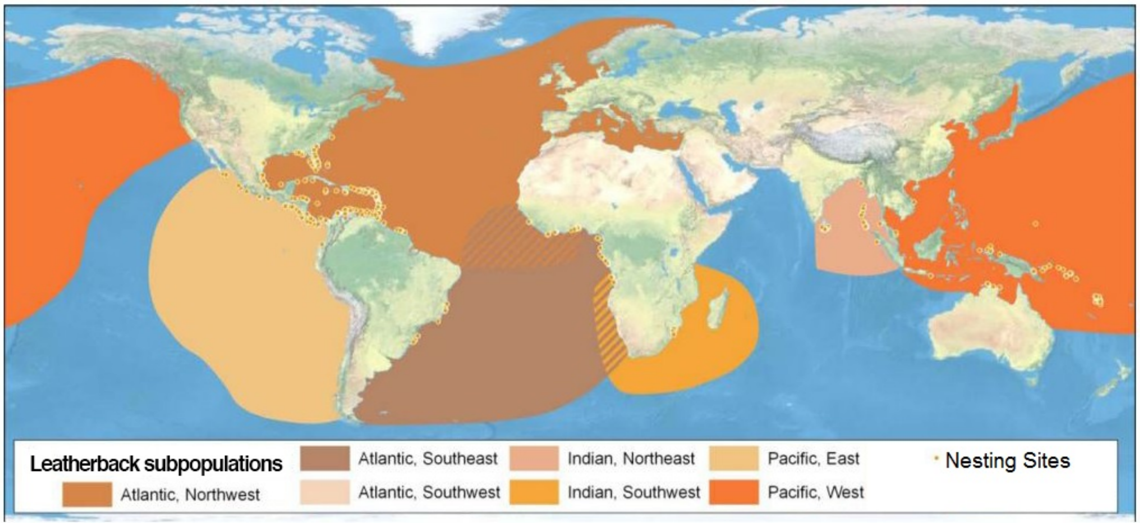


Figure 6. Map identifying the range of the endangered leatherback sea turtle (adapted from Wallace et al. 2013)

Life History

Age at maturity has been difficult to ascertain, with estimates ranging from 5 to 29 years (Spotila et al. 1996; Avens et al. 2009). Females lay up to seven clutches per season, with more than 65 eggs per clutch and eggs weighing >80 g (Wallace et al. 2007; Reina et al. 2002). 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). Females nest every 1 – 7 years. Natal homing, at least within an ocean basin, results in reproductive isolation between five broad geographic regions: eastern and western Pacific, eastern and western Atlantic, and Indian Ocean. Leatherback sea turtles migrate long, transoceanic distances between their tropical nesting beaches and the highly productive temperate waters where they forage, primarily on jellyfish and tunicates. These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight. Leatherbacks weigh ~33 percent more on their foraging grounds than at nesting, indicating that they probably catabolize fat reserves to fuel migration and subsequent reproduction (James et al. 2005; Wallace et al. 2006). Sea turtles must meet an energy threshold before returning to nesting beaches. Therefore, their remigration intervals (the time between nesting) are dependent upon foraging success and duration (Hays 2000; Price et al. 2004).

Abundance

Leatherbacks are globally distributed, with nesting beaches in the Pacific, Atlantic, and Indian oceans. Detailed population structure is unknown, but is likely dependent upon nesting beach location. Based on estimates calculated from nest count data, there are between 34,000 and 94,000 adult leatherbacks in the North Atlantic (TEWG 2007). In contrast, leatherback populations in the Pacific are much lower. Overall, Pacific populations have declined from an estimated 81,000 individuals to less than 3,000 total adults and subadults (Spotila et al. 2000).

Population abundance in the Indian Ocean is difficult to assess due to lack of data and inconsistent reporting. Available data from southern Mozambique show that approximately 10 females nest per year from 1994-2004, and about 296 nests per year counted in South Africa (NMFS 2013a).

Population Growth Rate

Population growth rates for leatherback sea turtles vary by ocean basin. Counts of leatherbacks at nesting beaches in the western Pacific indicate that the subpopulation has been declining at a rate of almost 6% per year since 1984 (Tapilatu et al. 2013). Leatherback subpopulations in the Atlantic Ocean however are showing signs of improvement. Nesting females in South Africa are increasing at an annual rate of 4 to 5.6%, and from 9 to 13% in Florida and the U.S. Virgin Islands (TEWG 2007), believed to be a result of conservation efforts.

Genetic Diversity

Analyses of mitochondrial DNA from leatherback sea turtles indicates a low level of genetic diversity, pointing to possible difficulties in the future if current population declines continue (Dutton et al. 1999). Further analysis of samples taken from individuals from rookeries in the Atlantic and Indian oceans suggest that each of the rookeries represent demographically independent populations (NMFS 2013a).

Distribution

Leatherback sea turtles are distributed in oceans throughout the world (Figure 6). Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Shoop and Kenney 1992). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Benson et al. 2011).

Status

The status of the Atlantic leatherback population has 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 has been a result of inconsistent beach and aerial surveys, cycles of erosion, and reformation of nesting beaches in the Guianas (representing the largest nesting area). Leatherbacks also show a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species. Coordinated efforts of data collection and analyses by the leatherback TEWG have helped to clarify the understanding of the Atlantic population status (TEWG 2007).

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 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). Nesting data for the Northern Caribbean stock is available from Puerto Rico, St. Croix (USVI), 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). 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). The West African nesting stock of leatherbacks is large and important, but it is a mostly unstudied aggregation. Two other small but growing stocks nest on the beaches of Brazil and South Africa.

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). The TEWG (2007) also determined that at of the time of their publication, leatherback sea turtle populations in the Atlantic were all stable or increasing with the exception of the Western Caribbean and West Africa populations. The latest review by NMFS and USFWS (2013b) suggests the leatherback nesting population is stable in most nesting regions of the Atlantic Ocean.

Critical Habitat

On March 23, 1979, leatherback critical habitat was identified adjacent to Sandy Point, St. Croix, Virgin Islands from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W (44 FR 17710). This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity; however, studies do not support significant critical habitat deterioration.

On January 20, 2012, NMFS issued a final rule to designate additional critical habitat for the leatherback sea turtle (50 C.F.R. 226). This designation includes approximately 43,798 km^2 stretching along the California coast from Point Arena to Point Arguello east of the 3000 m depth contour; and 64,760 km^2 stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 m depth contour. The designated areas comprise approximately 108558 km^2 of marine habitat and include waters from the ocean surface down to a maximum depth of 80 m. They were designated specifically because of the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (i.e., jellyfish), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

Recovery Goals

See the 1998 and 1991 Recovery Plans for the U.S. Pacific (USFWS and NMFS 1998b) and U.S Caribbean, Gulf of Mexico and Atlantic (NMFS and USFWS 1991) leatherback sea turtles for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top five recovery actions identified to support in the Leatherback 5-Year Action Plan:

1. Reduce fisheries interactions
2. Improve nesting beach protection and increase reproductive output
3. International cooperation
4. Monitoring and research
5. Public engagement

6.2.2.3 Status of Hawksbill Sea Turtles

The species was first listed under the Endangered Species Conservation Act (35 FR 8491) and listed as endangered under the ESA since 1973.

Species Description and Life History

The hawksbill turtle has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans (Figure 7).

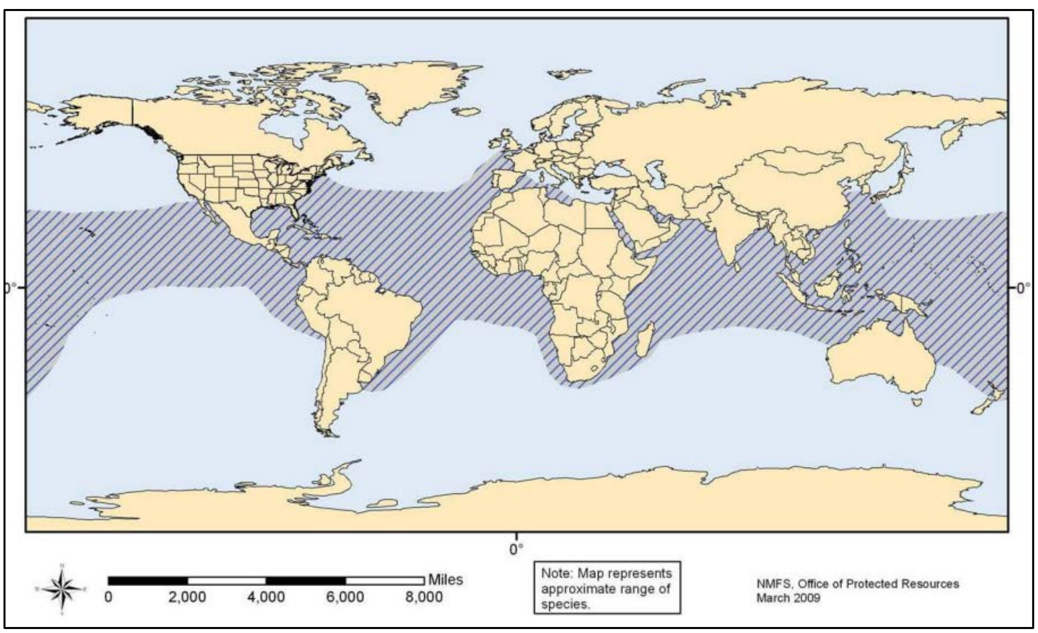


Figure 7. Map identifying the range of the endangered hawksbill sea turtle (http://www.nmfs.noaa.gov/pr/pdfs/rangemaps/hawksbill_turtle.pdf)

Hawksbill sea turtles reach sexual maturity at 20 – 40 years of age. Females return to their natal beaches every 2 – 5 years to nest (an average of 3 – 5 times per season). Clutch sizes are large (up to 250 eggs). Sex determination is temperature dependent, with warmer incubation producing more females. Hatchlings migrate to and remain in pelagic habitats until they reach approximately 22 – 25 cm in straight carapace length. As juveniles, they take up residency in coastal waters to forage and grow. As adults, hawksbills use their sharp beak-like mouths to feed on sponges and corals. Hawksbill sea turtles are highly migratory and use a wide range of habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Satellite tagged turtles have shown significant variation in movement and migration patterns. Distance traveled between nesting and foraging locations ranges from a few hundred to a few thousand km (Horrocks et al. 2001; Miller 1998).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section is broken down into: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the hawksbill sea turtle.

Abundance

Surveys at 88 nesting sites worldwide indicate that 22,004 – 29,035 females nest annually (NMFS and USFWS 2013a). In general, hawksbills are doing better in the Atlantic and Indian Ocean than in the Pacific Ocean, where despite greater overall abundance, a greater proportion of the nesting sites are declining.

Population Growth Rate

From 1980 to 2003, the number of nests at three primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) increased 15 percent annually (Heppell et al. 2005); however, due to recent declines in nest counts, decreased survival at other life stages, and updated population modeling, this rate is not expected to continue (NMFS and USFWS 2013a).

Genetic Diversity

Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor. Genetic analysis of hawksbill sea turtles foraging off the Cape Verde Islands identified three closely-related haplotypes in a large majority of individuals sampled that did not match those of any known nesting population in the western Atlantic, where the vast majority of nesting has been documented (Monzón-Argüello et al. 2010). Hawksbills in the Caribbean seem to have dispersed into separate populations (rookeries) after a bottleneck roughly 100,000-300,000 years ago (Leroux et al. 2012).

Distribution

The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific Oceans. In their oceanic phase, juvenile hawksbills can be found in *Sargassum* mats; post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997).

Status

Long-term data on the hawksbill sea turtle indicate that 63 sites have declined over the past 20 to 100 years (historic trends are unknown for the remaining 25 sites). Recently, 28 sites (68 percent) have experienced nesting declines, 10 have experienced increases, three have remained stable, and 47 have unknown trends. The greatest threats to hawksbill sea turtles are overharvesting of turtles and eggs, degradation of nesting habitat, and fisheries interactions. Adult hawksbills are harvested for their meat and carapace, which is sold as tortoiseshell. Eggs are taken at high levels, especially in Southeast Asia where collection approaches 100 percent in some areas. In addition, lights on or adjacent to nesting beaches are often fatal to emerging hatchlings and alters the behavior of nesting adults. The species' resilience to additional perturbation is low.

Critical Habitat

NMFS designated critical habitat for hawksbill sea turtles on September 2, 1998 around Mona and Monito Islands, Puerto Rico, which is outside the action area for this consultation.

Recovery Goals

The 1992 and 1998 Recovery Plans for the U.S. Caribbean, Atlantic and Gulf of Mexico (NMFS and USFWS 1993), and U.S. Pacific (USFWS and NMFS 1998a) populations of hawksbill sea turtles, respectively, contain complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top recovery actions identified to support in the Recovery Plans:

- Identify important nesting beaches
- Ensure long-term protection and management of important nesting beaches
- Protect and manage nesting habitat; prevent the degradation of nesting habitat caused by seawalls, revetments, sand bags, other erosion-control measures, jetties and breakwaters
- Identify important marine habitats; protect and manage populations in marine habitat
- Protect and manage marine habitat; prevent the degradation or destruction of important [marine] habitats caused by upland and coastal erosion
- Prevent the degradation of reef habitat caused by sewage and other pollutants
- Monitor nesting activity on important nesting beaches with standardized index surveys
- Evaluate nest success and implement appropriate nest-protection on important nesting beaches

- Ensure that law-enforcement activities prevent the illegal exploitation and harassment of sea turtles and increase law-enforcement efforts to reduce illegal exploitation
- Determine nesting beach origins for juveniles and subadult populations

6.2.3 Nassau Grouper

NMFS listed the Nassau grouper as threatened under the ESA effective July 29, 2016 (81 FR 42268, June 29, 2016).

Species Description and Life History

The Nassau grouper, *Epinephelus striatus* ((NMFS 2013b), is a moderate-sized serranid fish. As with many serranids, the Nassau grouper is slow-growing and long-lived; estimates range up to a maximum of 29 years (Bush et al. 1996). Using length-frequency analysis, which tends to exclude younger animals, a theoretical maximum age at 95 percent asymptotic size is 16 years. Individuals of more than 12 years of age are not common in fisheries, with more heavily fished areas yielding much younger fish on average. Most studies indicate a rapid growth rate for juveniles, which has been estimated to be about 10 mm/month total length (TL) for small juveniles, and 8.4-11.7 mm/month TL for larger juveniles (Beets and Hixon 1994; Eggleston 1995) . Maximum size is about 122 cm TL and maximum weight is about 25 kg (Humann and DeLoach 2002; Heemstra 1993; Froese 2010). Generation time (the interval between the birth of an individual and the subsequent birth of its first offspring) is estimated as 9-10 years (Sadovy and Eklund 1999). Male and female Nassau groupers reach sexual maturity at lengths between 40 and 45 centimeters (cm) standard length, about four to five years old. It is thought that sexual maturity is more determined by size, rather than age. Otolith studies indicate that the minimum age at maturity is between four and eight years; most groupers have spawned by age seven (Bush et al. 2006). Nassau groupers live to a maximum of 29 years.

Nassau groupers spawn once a year in large aggregations, in groups of a few dozen to thousands spawning at once. Nassau groupers move in groups towards the spawning aggregation sites parallel to the coast or along the shelf edge at depths between 20 and 33 m. Spawning runs occur in late fall through winter (i.e., a month or two before spawning is likely). Sea surface temperature is thought to be a key factor in the timing of spawning, with spawning occurring at waters temperatures between 25 and 26 degrees Celsius. Spawning aggregation sites are located near significant geomorphological features, such as reef projections (as close as 50 m to shore) and close to a drop-off into deep water over a wide depth range (six to sixty m). Sites are usually several hundred meters in diameter, with soft corals, sponges, stony coral outcrops, and sandy depressions. Nassau groupers stay on the spawning site for up to three months, spawning at the full moon or between the new and full moons. Spawning occurs within twenty minutes of sunset over the course of several days. There have been about fifty known spawning sites in insular areas throughout the Caribbean; many of these aggregations no longer form. Current spawning locations are found in Mexico, Bahamas, Belize, Cayman Islands, the Dominican Republic, Cuba, Puerto Rico and the USVI.

Fertilized eggs are transported offshore by ocean currents. Thirty-five to forty days after hatching, larvae recruit from oceanic environment to demersal habitats (at a size of about 32 mm TL). Juveniles inhabit macroalgae, coral clumps, and seagrass beds, and are relatively solitary. As they grow, they occupy progressively deeper areas and offshore reefs, where they may form schools of up to forty individuals. When not spawning, adults are most commonly found in waters less than one hundred meters deep. Nassau grouper diet changes with age. Juveniles eat plankton, pteropods, amphipods, and copepods. Adults are unspecialized piscivores, bottom-dwelling ambush suction predators (NMFS 2013b).

Distribution

The Nassau grouper's confirmed distribution currently includes "Bermuda and Florida (USA), throughout the Bahamas and Caribbean Sea" (e.g., Heemstra 1993). The occurrence of Nassau grouper from the Brazilian coast south of the equator as reported in Heemstra (1993) is "unsubstantiated" (Craig et al. 2011). The Nassau grouper has been documented in the Gulf of Mexico, at Arrecife Alacranes (north of Progreso) to the west off the Yucatan Peninsula, Mexico (Hildebrand et al. 1964). Nassau grouper is generally replaced ecologically in the eastern Gulf by red grouper (*E. morio*) in areas north of Key West or the Tortugas (Smith 1971). They are considered a rare or transient species off Texas in the northwestern Gulf of Mexico (Gunter and Knapp 1951; in Hoese and Moore 1998). The first confirmed sighting of Nassau grouper in the Flower Garden Banks National Marine Sanctuary, which is located in the northwest Gulf of Mexico approximately 180 km southeast of Galveston, Texas, was reported by (Foley et al. 2007). Many earlier reports of Nassau grouper up the Atlantic coast to North Carolina have not been confirmed. The Biological Report (Hill and Sadovy de Mitcheson 2013) provides a detailed description of the distribution, summarized in Figure 8.

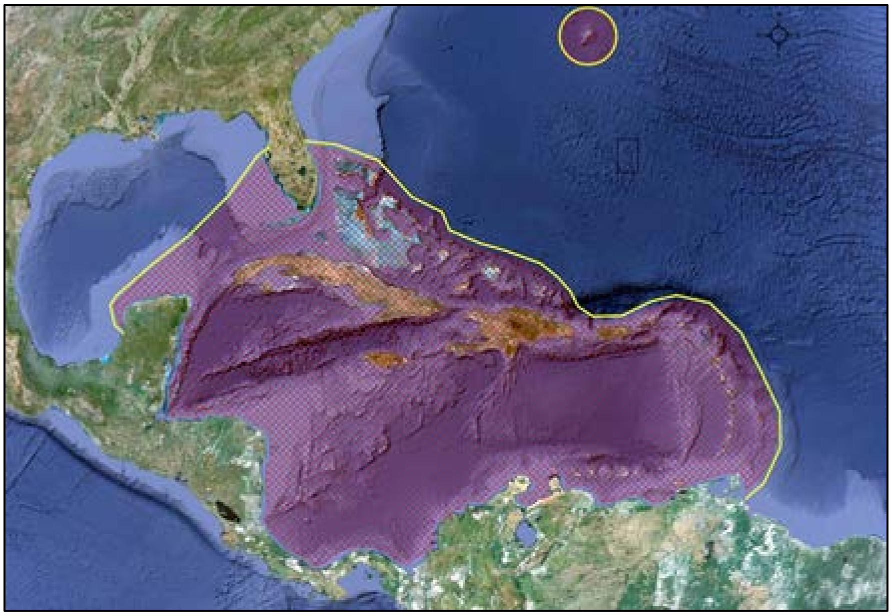


Figure 8. Range of Nassau grouper (*Epinephelus striatus*)

Population Dynamics

There is no range-wide abundance estimate available for Nassau grouper. The species is characterized as having patchy abundance due largely to differences in habitat availability or quality, and differences in fishing pressure in different locations (81 FR 42268). Although abundance has been reduced compared to historical levels, spawning still occurs and abundance is increasing in some locations, such as the Cayman Islands and Bermuda.

There is no population growth rate available for Nassau grouper. However, the available information from observations of spawning aggregations has shown steep declines (Aguilar-Perera 2006; Sala et al. 2001; Claro and Lindeman 2003). Some aggregation sites are comparatively robust and showing signs of increase (Whaylen et al. 2004; Vo et al. 2014).

Recent studies on Nassau grouper genetic variation has found strong genetic differentiation across the Caribbean subpopulations, likely due to barriers created by ocean currents and larval behavior (Jackson et al. 2014a).

Nassau grouper is distributed throughout the Caribbean, south to the northern coast of South America (Figure 8). Current Nassau grouper distribution is considered equivalent to its historical range, although abundance has been severely depleted.

Status

Historically, tens of thousands of Nassau grouper spawned at aggregation sites throughout the Caribbean. Since grouper species were reported collectively in landings data, it is not possible to know how many Nassau grouper were harvested, or estimate historic abundance. That these large spawning aggregations occurred in predictable locations at regular times made the species susceptible to over-fishing and was a cause of its decline. At some sites (e.g., Belize), spawning aggregations have decreased by over 80 percent in the last 25 years (Sala et al. 2001), or have disappeared entirely (e.g., Mexico; Aguilar-Perera 2006). Nassau groupers are also targeted for fishing throughout the year during non-spawning months. In some locations, spawning aggregations are increasing. Many Caribbean countries have banned or restricted Nassau grouper harvest, and it is believed that the areas of higher abundance are correlated with effective regulations (81 FR 42268). Because Nassau groupers are dependent upon coral reefs at various points in their life history, loss of coral reef habitat due to climate change will affect the abundance and distribution of the species. Increasing water temperatures may change the timing and location of spawning. Habitat degradation due to water pollution also poses a threat to the species. Nassau grouper populations have been reduced from historic abundance levels, and remain vulnerable to unregulated harvest, especially the spawning aggregations. NMFS determined that the species warrants listing as threatened.

Critical Habitat

No critical habitat has been designated for the Nassau grouper.

Recovery Goals

NMFS has not prepared a recovery plan for the Nassau grouper.

6.2.4 General Threats Faced by ESA-Listed Corals

Corals face numerous natural and man-made threats that shape their status and affect their ability to recover. Because many of the threats are the same or similar in nature for all listed coral species, those identified in this section are discussed in a general sense for all corals. All threats are expected to increase in severity in the future. More detailed information on the threats to listed corals is found in the Final Listing Rule (79 FR 53851; September 10, 2014). Threat information specific to a particular species is then discussed in the corresponding status sections where appropriate.

Several of the most important threats contributing to the extinction risk of corals are related to global climate change, which are discussed further in Section 7.2.1.

Ocean Warming

Ocean warming is one of the most important threats posing extinction risks to the listed coral species, but individual susceptibility varies among species. The primary observable coral response to ocean warming is bleaching of adult coral colonies, wherein corals expel their symbiotic algae in response to stress. For many corals, an episodic increase of only 1°C–2°C above the normal local seasonal maximum ocean temperature can induce bleaching. Corals can withstand mild to moderate bleaching; however, severe, repeated, and/or prolonged bleaching can lead to colony death. Coral bleaching patterns are complex, with several species exhibiting seasonal cycles in symbiotic algae density. Thermal stress has led to bleaching and mass mortality in many coral species during the past 25 years.

In addition to coral bleaching, other effects of ocean warming can harm virtually every life-history stage in reef-building corals. Impaired fertilization, developmental abnormalities, mortality, impaired settlement success, and impaired calcification of early life phases have all been documented. Average seawater temperatures in reef-building coral habitat in the wider Caribbean have increased during the past few decades and are predicted to continue to rise between now and 2100. Further, the frequency of warm-season temperature extremes (warming events) in reef-building coral habitat has increased during the past 2 decades and is predicted to continue to increase between now and 2100.

Ocean Acidification

Ocean acidification is a result of global climate change caused by increased CO_2 in the atmosphere that results in greater releases of CO_2 that is then absorbed by seawater. Reef-building corals produce skeletons made of the aragonite form of calcium carbonate. Ocean acidification reduces aragonite concentrations in seawater, making it more difficult for corals to build their skeletons. Ocean acidification has the potential to cause substantial reduction in coral calcification and reef cementation. Further, ocean acidification affects adult growth rates and fecundity, fertilization, pelagic planula settlement, polyp development, and juvenile growth. Ocean acidification can lead to increased colony breakage, fragmentation, and mortality. Based on observations in areas with naturally low pH, the effects of increasing ocean acidification may also include reductions in coral size, cover, diversity, and structural complexity.

As CO_2 concentrations increase in the atmosphere, more CO_2 is absorbed by the oceans, causing lower pH and reduced availability of calcium carbonate. Because of the increase in CO_2 and other GHGs 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. Along with ocean warming and disease, we consider ocean acidification to be one of the most important threats posing extinction risks to coral species between now and the year 2100, although individual susceptibility varies among the listed corals.

Diseases

Disease adversely affects various coral life history events by, among other processes, causing adult mortality, reducing sexual and asexual reproductive success, and impairing colony growth. A diseased state results from a complex interplay of factors including the cause or agent (e.g., pathogen, environmental toxicant), the host, and the environment. All coral disease impacts are presumed to be attributable to infectious diseases or to poorly described genetic defects. Coral disease often produces acute tissue loss. Other forms of "disease" in the broader sense, such as temperature-caused bleaching, are discussed in other threat sections (e.g., ocean warming because of climate change).

Coral diseases are a common and significant threat affecting most or all coral species and regions to some degree, although the scientific understanding of individual disease causes in corals remains very poor. The incidence of coral disease appears to be expanding geographically, though the prevalence of disease is highly variable between sites and species. Increased prevalence and severity of diseases is correlated with increased water temperatures, which may correspond to increased virulence of pathogens, decreased resistance of hosts, or both. Moreover, the expanding coral disease threat may result from opportunistic pathogens that become damaging only in situations where the host integrity is compromised by physiological stress or immune suppression. Overall, there is mounting evidence that warming temperatures and coral bleaching responses are linked (albeit with mixed correlations) with increased coral disease prevalence and mortality.

Trophic Effects of Reef Fishing

Fishing, particularly overfishing, can have large-scale, long-term ecosystem-level effects that can change ecosystem structure from coral-dominated reefs to algal-dominated reefs (“phase shifts”). Even fishing pressure that does not rise to the level of overfishing potentially can alter trophic interactions that are important in structuring coral reef ecosystems. These trophic interactions include reducing population abundance of herbivorous fish species that control algal growth, limiting the size structure of fish populations, reducing species richness of herbivorous fish, and releasing corallivores from predator control.

In the Caribbean, parrotfishes can graze at rates of more than 150,000 bites per square meter (m^2) per day (Carpenter 1986), and thereby remove up to 90-100 percent of the daily primary production (e.g., algae; Hatcher 1997). With substantial populations of herbivorous fishes, as long as the cover of living coral is high and resistant to mortality from environmental changes, it is very unlikely that the algae will take over and dominate the substrate. However, if herbivorous fish populations, particularly large-bodied parrotfish, are heavily fished and a major mortality of coral colonies occurs, then algae can grow rapidly and prevent the recovery of the coral population. The ecosystem can then collapse into an alternative stable state, a persistent phase shift in which algae replace corals as the dominant reef species. Although algae can have negative effects on adult coral colonies (e.g., overgrowth, bleaching from toxic compounds), the ecosystem-level effects of algae are primarily from inhibited coral recruitment. Filamentous algae can prevent the colonization of the substrate by planula larvae by creating sediment traps that obstruct access to a hard substrate for attachment. Additionally, macroalgae can block successful colonization of the bottom by corals because the macroalgae takes up the available space and causes shading, abrasion, chemical poisoning, and infection with bacterial disease. Trophic effects of fishing are a medium importance threat to the extinction risk for listed corals.

Sedimentation

Human activities in coastal and inland watersheds introduce sediment into the ocean by a variety of mechanisms including river discharge, surface runoff, groundwater seeps, and atmospheric deposition. Humans also introduce sewage into coastal waters through direct discharge, treatment plants, and septic leakage. Elevated sediment levels are generated by poor land use practices and coastal and nearshore construction.

The most common direct effect of sedimentation is sediment landing on coral surfaces as it settles out from the water column. Corals with certain morphologies (e.g., mounding) can passively reject settling sediments. In addition, corals can actively remove sediment but at a significant energy cost. Corals with large calices (skeletal component that holds the polyp) tend to be better at actively rejecting sediment. Some coral species can tolerate complete burial for several days. Corals that cannot remove sediment will be smothered and die. Sediment can also cause sublethal effects such as reductions in tissue thickness, polyp swelling, zooxanthellae loss, and excess mucus production. In addition, suspended sediment can reduce the amount of light in the water column, making less energy available for coral photosynthesis and growth. Sedimentation also impedes fertilization of spawned gametes and reduces larval settlement and survival of recruits and juveniles.

Nutrient Enrichment

Elevated nutrient concentrations in seawater affect corals through two main mechanisms: direct impacts on coral physiology, and indirect effects through stimulation of other community components (e.g., macroalgal turfs and seaweeds, and filter feeders) that compete with corals for space on the reef. Increased nutrients can decrease calcification; however, nutrients may also enhance linear extension while reducing skeletal density. Either condition results in corals that are more prone to breakage or erosion, but individual species do have varying tolerances to increased nutrients. Anthropogenic nutrients mainly come from point-source discharges (such as rivers or sewage outfalls) and surface runoff from modified watersheds. Natural processes, such as *in situ* nitrogen fixation and delivery of nutrient-rich deep water by internal waves and upwelling, also bring nutrients to coral reefs.

6.2.5 Status of ESA-Listed Corals

6.2.5.1 Elkhorn Coral (Acropora palmata)

Elkhorn coral was listed as threatened under the ESA in 2006.

Species Description and Life History

Elkhorn coral occurs throughout coastal areas in the Caribbean, Gulf of Mexico, and southwestern Atlantic (Figure 9).

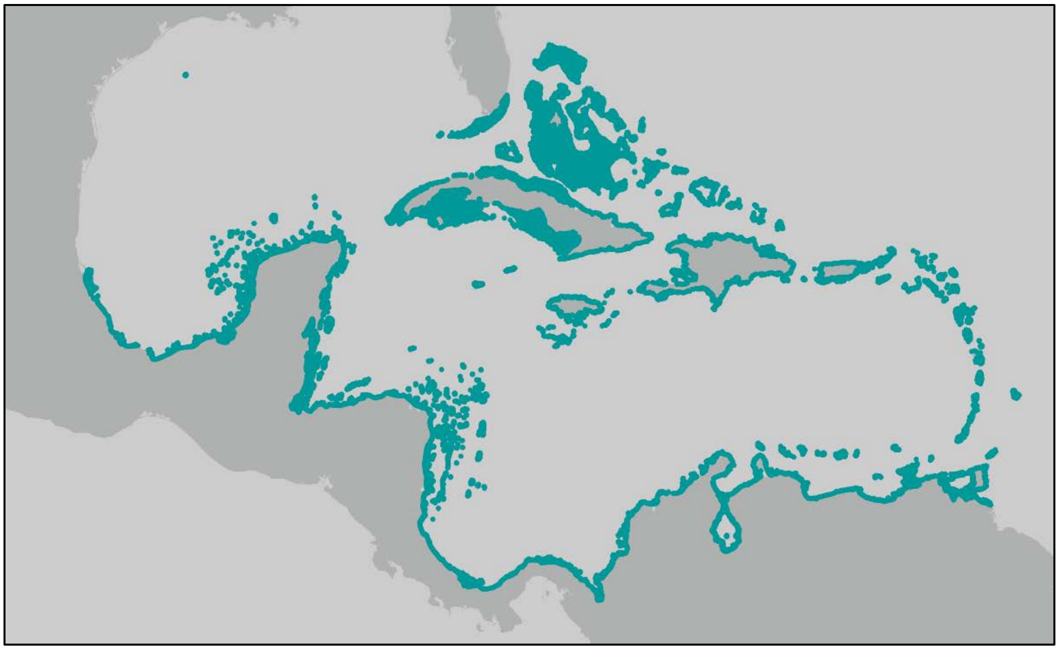


Figure 9. Map showing range of elkhorn and staghorn corals

Elkhorn corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, elkhorn coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005).

Along with staghorn coral, elkhorn coral is the only other large, branching species of coral to produce and occupy vast complex environments within the Caribbean Sea's reef system. In all, there appears to be two distinct populations of elkhorn coral, a western Caribbean population and an eastern (Baums et al. 2005) based on genetic analyses.

Elkhorn coral, like most stony corals, employ both sexual and asexual reproductive strategies to propagate. Sexual reproduction in corals includes gametogenesis, the process in which cells undergo meiosis to form gametes within the polyps. Since elkhorn coral is hermaphroditic, each polyp contains both sperm and egg cells that are released together in a "bundle", causing the coral gametes to develop externally from the parental colony. Elkhorn coral reproduces sexually after the full moon of July, August, and/or September, depending on location and timing of the full moon (*Acropora* Biological Review Team 2005). Split spawning (spawning over a 2-month period) has been reported from the Florida Keys Fogarty et al. (2012). The estimated size at sexual maturity is approximately 250 in^2 (1,600 square cm [cm^2]), and growing edges and encrusting base areas are not fertile (Soong and Lang 1992). Larger colonies have higher fecundity per unit area, as do the upper branch surfaces (Soong and Lang 1992). Although selffertilization is possible, elkhorn coral is largely self-incompatible (Baums et al. 2005; Fogarty et al. 2012). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies in the field. Rates of post-settlement mortality after nine months are high based on settlement experiments (Szmant and Miller 2005).

Reproduction occurs primarily through asexual reproduction, generating multiple genetically identical colonies. Elkhorn coral can quickly monopolize large spaces of shallow ocean floor through fragment dissemination. A branch of elkhorn coral can be carried by waves and currents away from the mother colony to distances that range from 0.1 – 100 m, but fragments usually travel less than 30 m (NMFS 2005).

Because large colonies of elkhorn coral contain several thousand partially autonomous polyps, growth rates for the species are conveyed through the measurement of linear extensions of the organisms' skeletal branches. Depending on the size and location of the colony, physical growth rates for elkhorn corals range from approximately four to eleven cm per year. Branches are up to approximately 50 cm wide and range in thickness of about 4 - 5 cm. Individual colonies can grow to at least 2 m in height and 4 m in diameter (NMFS 2005). Total lifespan for the species is unknown (NMFS 2014).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the elkhorn coral.

Genetic samples from 11 locations throughout the Caribbean indicate that elkhorn coral populations in the eastern Caribbean (St. Vincent and the Grenadines, U.S. Virgin Islands, Curaçao, and Bonaire) have had little or no genetic exchange with populations in the western Atlantic and western Caribbean (Bahamas, Florida, Mexico, Panama, Navassa, and Puerto Rico; Baums et al. 2005). While Puerto Rico is more closely connected with the western Caribbean, it is an area of mixing with contributions from both regions (Baums et al. 2005). Models suggest that the Mona Passage between the Dominican Republic and Puerto Rico promotes dispersion of larval and gene flow between the eastern Caribbean and western Caribbean (Baums et al. 2006).

Colonial species present a special challenge in determining the appropriate unit to evaluate for abundance. However, the present population of Elkhorn coral is continuing at a very low abundance due to large declines in the past several decades (NMFS 2005). Genetically depauperate populations with lower densities (0.13 ± 0.08 colonies per m^2) characterize the western Caribbean. The eastern Caribbean populations are characterized by denser (0.30 ± 0.21 colonies per m^2), genotypically richer stands (Baums et al. 2006).

Baums et al. (2006) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment. The research team claims that the postulated geographic differences in the contribution of reproductive modes to population structure may be related to habitat characteristics, possibly the amount of shelf area available.

Genotypic diversity is highly variable for elkhorn coral. From the survey data, it can be inferred that genetic variability is more common in colonies within eastern populations as opposed to western. At two sites in the Florida Keys, only one genotype per site was detected out of 20 colonies sampled at each site (Baums et al. 2005). In contrast, sites within the eastern Caribbean displayed high variability. All 15 colonies sampled in Navassa had unique genotypes (Baums et al. 2006). Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotypes out of 120 samples; Zubillaga et al. 2008) and in Bonaire and Curaçao (18 genotypes of 22 samples and 19 genotypes of 20 samples, respectively; Baums et al. 2006). In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24 and 65 percent of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006). A more-recent survey conducted along the coast of Puerto Rico found unique genotypes in 75 percent of the samples with high genetic diversity (Mège et al. 2014).

Elkhorn coral occurs in turbulent water on the back reef, fore reef, reef crest, and spur and groove zone in water ranging from one to thirty m in depth. Historically, elkhorn coral inhabited most waters of the Caribbean between one to five m depth. This included a diverse set of areas comprising of zones along Puerto Rico, Hispaniola, the Yucatan peninsula, the Bahamas, the southwestern Gulf of Mexico, the Florida Keys, the Southeastern Caribbean islands, and the northern coast of South America as seen in Figure 14 (Dustan and Halas 1987; Goreau 1959; Jaap 1984; Kornicker and Boyd 1962; Scatterday 1974; Storr 1964). While the present-day spatial distribution of elkhorn coral is similar to its historic spatial distribution, its presence within its range has become increasingly sparse due to declines in the latter half of the 20th century from a variety of abiotic and biotic threats.

There is some density data available for elkhorn corals in Florida, Puerto Rico, the USVI and Cuba. In Florida, elkhorn coral was detected at zero to 78 percent of the sites surveyed between 1999 and 2017. Average density ranged from 0.001 to 0.12 colonies per m^2 (NOAA, unpublished data). Elkhorn coral was encountered less frequently during benthic surveys in the USVI from 2002 to 2017. It was observed at zero to seven percent of surveyed reefs, and average density ranged from 0.001 to 0.01 colonies per m^2 (NOAA, unpublished data). Maximum elkhorn coral density at ten sites in St. John, USVI was 0.18 colonies per m^2 (Muller et al. 2014). In Puerto Rico, average density ranged from 0.002 to 0.09 colonies per m^2 in surveys conducted between 2008 and 2018, and elkhorn coal was observed on one to 27 percent of surveyed sites (NOAA, unpublished data). Density estimates from sites in Cuba range from 0.14 colonies per m^2 (Alcolado et al. 2010) to 0.18 colonies per m^2 (González-Díaz et al. 2010).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the USVI in 2017. Hurricane impacts included large, overturned, and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 45 to 77 percent of elkhorn corals were impacted (NOAA 2018a). Survey data for impacts to elkhorn corals are not available for the USVI or Florida, although qualitative observations indicate that damage was widespread but variable by site.

Based on population estimates from both the Florida Keys and St. Croix, USVI, there are at least hundreds of thousands of elkhorn coral colonies. Absolute abundance is higher than estimates from these two locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. Across the Caribbean, percent cover appears to have remained relatively stable, albeit it at extremely low levels, since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s, indicating potential decreases in the extent of occurrence and effects on the species' range. However, the proportions of Caribbean sites where elkhorn coral is present and dominant have recently stabilized since the mid-2000s. There are locations such as the U.S. Virgin Islands where populations of elkhorn coral appear stable or possibly increasing in abundance and some such as the Florida Keys where population number appears to be decreasing.

Status

The decline in the total abundance of elkhorn coral has been attributed to a series of stressors consisting of disease, temperature-induced bleaching, excessive sedimentation, nitrification, pollution(i.e. oxybenzone from sunscreen), and large hurricanes/tropical storms (Brainard et al. 2011; Downs et al. 2016; Hernandez-Delgado et al. 2011; Mayor et al. 2006; Rogers and Muller 2012). It is believed that these effects act synergistically with one another thereby increasing the overall damage to already-stressed elkhorn coral colonies that have undergone disturbance by another threat. The current population trend appears to be steady, although there are places where populations continue to decrease and others where there appears to be modest or contained recovery (Miller et al. 2013). However, even if growth and recruitment end up surpassing mortality, this species requires prompt analysis and monitoring on a regional scale. Reasoning for this includes the current presence of areas with low genetic diversity and density within western Caribbean populations along with localized high rates of disease and bleaching (Miller et al. 2013).

Critical Habitat

Critical habitat units for elkhorn and staghorn coral were designated in 2008 and include Florida (portions of Southeastern Florida and the Florida Keys), Puerto Rico, St. Thomas/St. John, and St. Croix. Elkhorn and staghorn coral critical habitat is described further in Section 7.2.7.

Recovery Goals

The 2015 Elkhorn Coral (*Acropora palmata*) and Staghorn Coral (*A. cervicornis*) Recovery Plan contains complete downlisting/delisting criteria for each of the two following recovery goals:

- Ensure population viability

    - Specific criteria include: 1) Preserving Abundance; 2) Maintaining Genotypic Diversity; and 3) Properly Observing and Recording Recruitment Rates
- Eliminate or sufficiently abate global, regional, and local threats
    - Specific criteria include: 1) Developing quantitative recovery criterion through research to identify, treat, and reduce outbreaks of coral disease; 2) Controlling the Local and Global Impacts of Rising Ocean Temperature and Acidification; 3) Reducing the Loss of Recruitment Habitat (if criterion 1, preserving abundance, is met then this objective is complete; 4) Reducing sources of nutrients, sediments, and contaminants; 5) Developing and adopting appropriate and effective regulatory mechanisms to abate threats; 6) Reducing impacts of natural and anthropogenic abrasion and breakage; and 7) Reducing impacts of predation.

6.2.5.2 Staghorn Coral (Acropora cervicornis)

Staghorn coral has the same range as elkhorn coral, occurring throughout coastal areas in the Caribbean, Gulf of Mexico, and southwestern Atlantic (Figure 9).

Species Description and Life History

Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges form 0.25 – 5 cm (Lirman et al. 2010), and linear branch growth rates have been reported to range between 3 – 11.5 cm per year (*Acropora* Biological Review Team 2005). The species can exist as isolated branches, individual colonies up to about 1.5 m diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (*Acropora* Biological Review Team 2005).

Staghorn coral naturally occurs on spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats (Cairns 1982; Davis 1982; Gilmore and Hall 1976; Goldberg 1973; Jaap 1984; Miller et al. 2008; Wheaton and Jaap 1988). Historically it grew in thickets in water ranging from approximately 5 – 20 m in depth; though it has rarely been found to approximately 60 m (Davis 1982; Jaap 1984; Jaap et al. 1989; Schuhmacher and Zibrowius 1985; Wheaton and Jaap 1988). At the northern extent of its range, it grows in deeper water, 16-30 m (Goldberg 1973). Historically, staghorn coral was one of the primary constructors of mid-depth 10-15 m reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatan peninsula (Adey 1978). In the Florida Keys, staghorn coral occurs in various habitats but is most prevalent on patch reefs as opposed to their former abundance in deeper fore-reef habitats (i.e., 5 - 22 m; Miller et al. 2008). There is no evidence of range constriction, though loss of staghorn coral at the reef level has occurred (*Acropora* Biological Review Team 2005).

Precht and Aronson (2004) suggest that coincident with climate warming, staghorn coral recently re-occupied its historic range after contracting to south of Miami, Florida, during the late Holocene. They based this idea on the presence of large thickets off Ft. Lauderdale, Florida, which were discovered in 1998 and had not been reported in the 1970s or 1980s (Precht and Aronson 2004). However, because the presence of sparse staghorn coral colonies in Palm Beach County, north of Ft. Lauderdale, was reported in the early 1970s (though no thicket formation was reported; Goldberg 1973), there is uncertainty associated with whether these thickets were present prior to their discovery or if they recently appeared coincident with warming. The proportion of reefs with staghorn coral present decreased dramatically after the Caribbean-wide mass mortality in the 1970s and 1980s, indicating the spatial structure of the species has been affected by extirpation from many localized areas throughout its range (Jackson et al. 2014a).

Staghorn coral was observed in 21 out of 301 stations between 2011 and 2013 in stratified random surveys designed to detect *Acropora* colonies along the south, southeast, southwest, and west coasts of Puerto Rico (García-Sais et al. 2013). Staghorn coral was also observed at 16 sites outside of the surveyed area. The largest colony was 60 cm and density ranged from one to ten colonies per fifteen m^2 (García-Sais et al. 2013).

Relative to other corals, staghorn coral has a high growth rate that has allowed acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates, measured as skeletal extension of the end of branches, range from approximately four to eleven cm per year (*Acropora* Biological Review Team 2005). Annual linear extension has been found to be dependent on the size of the colony. New recruits and juveniles typically grow at slower rates. Stressed colonies and fragments may also exhibit slower growth.

Staghorn coral is a hermaphroditic broadcast spawning species. The spawning season occurs several nights after the full moon in July, August, or September depending on location and timing of the full moon and may be split over the course of more than one lunar cycle (Szmant 1986; Vargas-Angel et al. 2006). The estimated size at sexual maturity is approximately seventeen cm branch length, and large colonies produce proportionally more gametes than small colonies (Soong and Lang 1992). Basal and branch tip tissue is not fertile (Soong and Lang 1992). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies. Laboratory studies have found that certain species of crustose-coralline algae produce exudates that facilitate larval settlement and post-settlement survival (Ritson-Williams et al.).

Reproduction occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Tunnicliffe 1981). The combination of branching morphology, asexual fragmentation, and fast growth rates, relative to other corals, can lead to persistence of large areas dominated by staghorn coral. The combination of rapid skeletal growth rates and frequent asexual reproduction by fragmentation can enable effective competition and can facilitate potential recovery from disturbances when environmental conditions permit. However, low sexual reproduction can lead to reduced genetic diversity and limits the capacity to repopulate spatially dispersed sites.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the staghorn coral.

Miller et al. (2013) extrapolated population abundance of staghorn coral in the Florida Keys and Dry Tortugas from stratified random samples across habitat types. Population estimates of staghorn coral in the Florida Keys were 10.2 ± 4.6 (standard error [SE]) million colonies in 2005, 6.9 ± 2.4 (SE) million colonies in 2007 and 10.0 ± 3.1 (SE) million colonies in 2012. Population estimates in the Dry Tortugas were 0.4 ± 0.4 (SE) million colonies in 2006 and 3.5 ± 2.9 (SE) million colonies in 2008, though the authors note their sampling scheme in the Dry Tortugas was not optimized for staghorn coral. Because these population estimates were based on random sampling, differences in abundance estimates between years is more likely to be a function of sample design rather than population trends. In both the Florida Keys and Dry Tortugas, most of the population was dominated by small colonies less than 12-in (30 cm) diameter. Further, partial mortality was reported as highest in 2005 with up to 80 percent mortality observed and lowest in 2007 with a maximum of 30 percent. In 2012, partial mortality ranged from 20-50 percent across most size classes.

Staghorn coral historically was one of the dominant species on most Caribbean reefs, forming large, single-species thickets and giving rise to the nominal distinct zone in classical descriptions of Caribbean reef morphology (Goreau 1959). Massive, Caribbean-wide mortality, apparently primarily from white band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events has added to the decline of staghorn coral (Brainard et al. 2011). In locations where quantitative data are available (Florida, Jamaica, USVI, Belize), there was a reduction of approximately 92 to greater than 97 percent between the 1970s and early 2000s (*Acropora* Biological Review Team 2005).

Since the 2006 listing of staghorn coral as threatened, continued population declines have occurred in some locations with certain populations of both listed *Acropora* species (staghorn and elkhorn) decreasing up to an additional 50 percent or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008). There are some small pockets of remnant robust populations such as in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005; Riegl et al. 2009), and Dominican Republic (Lirman et al. 2010). Additionally, Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 miles (70.2 km) of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold-water die-off of the 1970s. Cover of staghorn coral increased on a Jamaican reef from 0.6 percent in 1995 to 10.5 percent in 2004 (Idjadi et al. 2006).

Riegl et al. (2009) monitored staghorn coral in photo plots on the fringing reef near Roatan, Honduras from 1996 to 2005. Staghorn coral cover declined from 0.42 percent in 1996 to 0.14 percent in 1999 after the Caribbean bleaching event in 1998 and mortality from run-off associated with a Category 5 hurricane. Staghorn coral cover further declined to 0.09 percent in 2005. Staghorn coral colony frequency decreased 71 percent between 1997 and 1999. In sharp contrast, offshore bank reefs near Roatan had dense thickets of staghorn coral with 31 percent cover in photo-quadrats in 2005 and appeared to survive the 1998 bleaching event and hurricane, most likely due to bathymetric separation from land and greater flushing. Modeling showed that under undisturbed conditions, retention of the dense staghorn coral stands on the banks off Roatan is likely with a possible increased shift towards dominance by other coral species. However, the authors note that because their data and the literature seem to point to extrinsic factors as driving the decline of staghorn coral, it is unclear what the future may hold for this dense population (Riegl et al. 2009).

While cover of staghorn coral increased from 0.6 percent in 1995 to 10.5 percent in 2004 (Idjadi et al. 2006) and 44 percent in 2005 on a Jamaican reef, it collapsed after the 2005 bleaching event and subsequent disease to less than 0.5 percent in 2006 (Quinn and Kojis 2008). A cold water die-off across the lower to upper Florida Keys in January 2010 resulted in the complete mortality of all staghorn coral colonies at 45 of the 74 reefs surveyed (61 percent) (61 percent; Schopmeyer et al. 2012). Walker et al. (2012) report increasing size of 2 thickets (expansion of up to 7.5 times the original size of 1 of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50 percent highlighting the dynamic nature of staghorn coral distribution via fragmentation and re-attachment.

A report on the status and trends of Caribbean corals over the last century indicates that cover of staghorn coral has remained relatively stable (though much reduced) throughout the region since the large mortality events of the 1970s and 1980s. The frequency of reefs at which staghorn coral was described as the dominant coral has remained stable. The number of reefs with staghorn coral present declined during the 1980s (from approximately 50 to 30 percent of reefs), remained relatively stable at 30 percent through the 1990s, and decreased to approximately 20 percent of the reefs in 2000-2004 and approximately 10 percent in 2005-2011 (Jackson et al. 2014a).

Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from nine regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 500 km apart are genetically different from each other with low gene flow across the greater Caribbean. Fine-scale genetic differences have been detected at reefs separated by as little as two km, suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A. prolifera* with staghorn coral (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). Populations in Florida and Honduras are genetically distinct from each other and other populations in the USVI, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the USVI and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

Staghorn coral is distributed throughout the Caribbean Sea, in the southwestern Gulf of Mexico, and in the western Atlantic Ocean. The fossil record indicates that during the Holocene epoch, staghorn coral was present as far north as Palm Beach County in southeast Florida (Lighty et al. 1978), which is also the northern extent of its current distribution (Goldberg 1973).Staghorn coral commonly occurs in water ranging from five to twenty m in depth, though it occurs in depths of 16-30 m at the northern extent of its range, and has been rarely found to 60 m in depth.

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the USVI in 2017. At 153 survey locations in Puerto Rico, approximately 38 to 54 percent of staghorn coral colonies were impacted (NOAA 2018a). In a post-hurricane survey of 57 sites in Florida, all of the staghorn coral colonies encountered were damaged (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the USVI, though qualitative observations indicate that damage was also widespread but variable by site.

Based on population estimates, there are at least tens of millions of colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. There is no evidence of range constriction or extirpation at the island level. However the species is absent at the reef level. Populations appear to consist mostly of isolated colonies or small groups of colonies compared to the vast thickets once prominent throughout its range. Thickets are a prominent feature at only a few known locations. Across the Caribbean, percent cover appears to have remained relatively stable since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s. There are examples of increasing trends in some locations (Dry Tortugas and southeast Florida), but not over larger spatial scales or longer periods. Population model projections from Honduras at one of the only known remaining thickets indicate the retention of this dense stand under undisturbed conditions. If refuge populations are able to persist, it is unclear whether they would be able to repopulate nearby reefs as observed sexual recruitment is low. Thus, we conclude that the species has undergone substantial population decline and decreases in the extent of occurrence throughout its range. Percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable since the mid-1980s and since the listing of the species as threatened in 2006. We also conclude that population abundance is at least tens of millions of colonies, but likely to decrease in the future with increasing threats.

Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. Although localized mortality events have continued to occur, percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable over its range since the mid-1980s. There is evidence of synergistic effects of threats for this species where the effects of increased nutrients are combined with acidification and sedimentation. Staghorn coral is highly susceptible to a number of threats, and cumulative effects of multiple threats are likely to exacerbate vulnerability to extinction. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because staghorn coral is limited to areas with high, localized human impacts and predicted increasing threats. Staghorn coral commonly occurs in water ranging from five to twenty m in depth, though it occurs in depths of 16-30 m at the northern extent of its range and has been rarely found to 60 m in depth. It occurs in spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats. This habitat heterogeneity moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef and hard bottom environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Its absolute population abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined and is higher than the estimate from these two locations due to the occurrence of the species in many other areas throughout its range. Staghorn coral has low sexual recruitment rates, which exacerbates vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated. In contrast, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate the species' vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Critical Habitat

Elkhorn and staghorn coral critical habitat is described further in Section 7.2.7.

Recovery Goals

The recovery goals for elkhorn and staghorn corals were described in the 2015 Elkhorn Coral (*Acropora palmata*) and Staghorn Coral (*A. cervicornis*) Recovery Plan (NMFS 2015b) and detailed in Section 7.2.6.1 (Elkhorn Coral). Two recovery goals were identified for Atlantic acroporid corals:

- Ensure population viability
- Eliminate or sufficiently abate global, regional, and local threats.

6.2.5.3 Pillar Coral (Dendrogyra cylindrus)

On September 10, 2014, NMFS listed pillar star coral as threatened (79 FR 53851).

Species Description and Life History

Pillar coral is present in the western Atlantic Ocean and throughout the greater Caribbean Sea, though absent from the southwest Gulf of Mexico (Tunnell Jr. 1988; Figure 10).

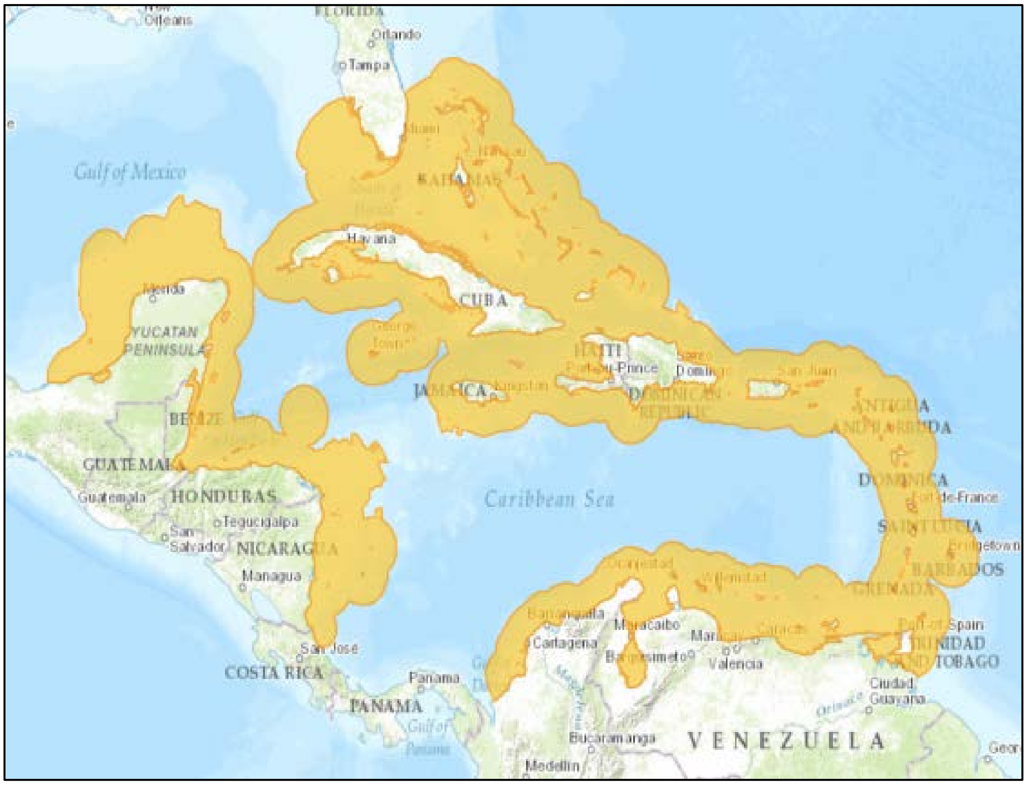


Figure 10. Range map for pillar coral (from Aronson et al. 2008a)

Pillar corals form tubular columns on top of encrusted foundations. Colonies are generally grey-brown in color and may reach approximately three m in height. Polyps' tentacles remain extended during the day, giving columns a furry appearance.

Brainard et al. (2011) identified a single known colony in Bermuda that is in poor condition. There is fossil evidence of the presence of the species off Panama less than 1,000 years ago, but it has been reported as absent today (FFWCC 2013). Pillar coral inhabits most reef environments in water depths ranging from approximately one to twenty-five m, but it is most common in water between approximately five to fifteen m deep (Acosta and Acevedo 2006; Cairns 1982; Goreau and Wells 1967).

Reported average growth rates for pillar coral have been documented to be approximately 1.8-2.0 cm per year in linear extension within the Florida Keys, compared to 0.8 cm per year as reported in Colombia and Curaçao. Partial mortality rates are size-specific with larger colonies having greater rates. Frequency of partial mortality can be high (e.g., 65 percent of 185 colonies surveyed in Colombia), while the amount of partial mortality per colony is generally low (average of three percent of tissue area affected per colony).

Pillar coral is a gonochoric broadcast spawning species with relatively low annual egg production for its size. The combination of gonochoric spawning with persistently low population densities is expected to yield low rates of successful fertilization and low larval supply. Sexual recruitment of this species is low, and reports indicate juvenile colonies are lacking in the Caribbean. Spawning has been observed to occur several nights after the full moon of August in the Florida Keys (Neely et al. 2013; Waddell and Clarke 2008) and in La Parguera, Puerto Rico (Szmant 1986). Pillar coral can also reproduce asexually by fragmentation following storms or other physical disturbance, but it is uncertain how much storm-generated fragmentation contributes to asexually produced offspring.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the pillar coral.

Pillar coral is uncommon but conspicuous with scattered, isolated colonies and is rarely found in aggregations. Benthic cover is generally less than one percent in monitoring studies. Mean density of pillar coral was approximately 0.5 colonies per ten m^2 in the Florida Keys between 2005 and 2007. In a study of pillar coral demographics at Providencia Island, Colombia, 283 pillar coral colonies were detected in a survey of 1.66 square kilometers (km^2) for an overall density of approximately 450 colonies per square mile (mi^2).

Information on pillar coral is most extensive for Florida. Pillar coral ranked as the least abundant to third least abundant coral species in stratified random surveys of the Florida Keys between 2005 and 2009 and was not encountered in surveys in 2012 (Miller et al. 2013). Pillar coral was seen only on the ridge complex and mid-channel reefs at densities of approximately 1 and 0.1 colonies per 10 m^2 (approximately 100 ft^2), respectively, between 2005 and 2010 in surveys from West Palm Beach to the Dry Tortugas (Burman et al. 2012). In surveys conducted between 1999 and 2016 from Palm Beach to the Dry Tortugas, pillar coral was present at 2 percent of sites surveyed and ranged in density from 0 to 0.4 colonies per m^2 with an average density of 0.004 colonies per 10 m^2 (approximately 100 ft^2; NOAA National Coral Reef Monitoring Program [NCRMP]). In 2014, there were 714 known colonies of pillar coral along the Florida reef tract from southeast Florida to the Dry Tortugas. By 2017, many of these colonies had suffered tissue loss, and over half (57 percent) suffered complete mortality due to disease, most likely associated with multiple years of warmer than normal temperatures (Lewis et al. 2017); K. Neely and C. Lewis, Keys Marine Lab, unpublished data). The majority of these colonies were lost from the northern portion of the reef tract (Figure 11).

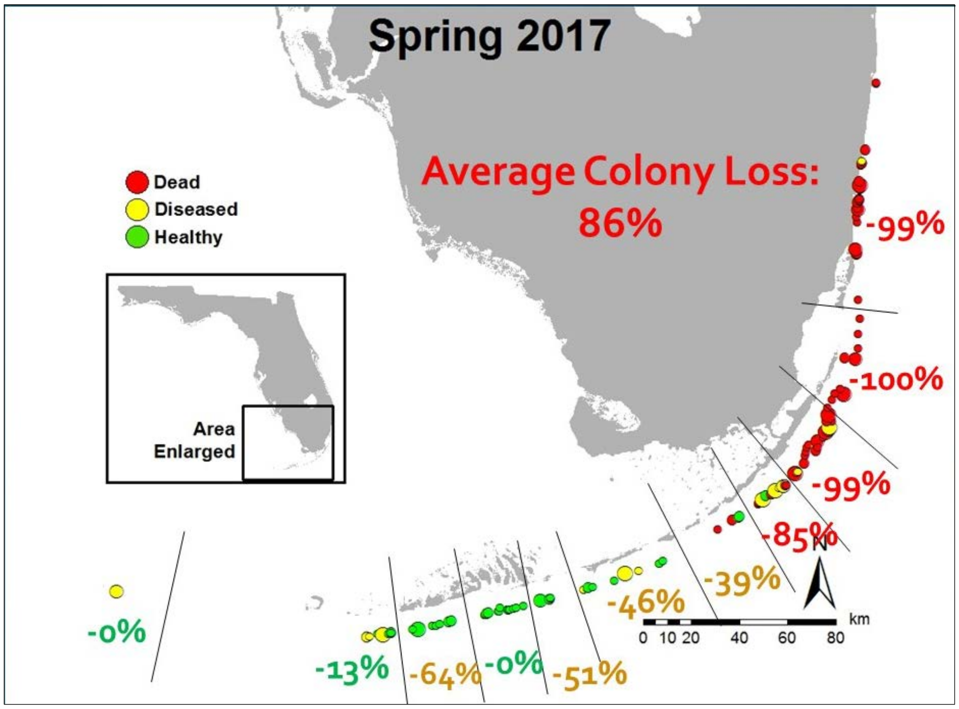


Figure 11. Condition of known pillar coral colonies in Florida between 2014 and 2017 (Figure courtesy of K. Neely and C. Lewis)

Density of pillar corals in other areas of the Caribbean is also low and on average less than 0.1 colonies per 10 m^2. The average number of pillar coral colonies in remote reefs off southwest Cuba was 0.013 ± 0.045 colonies per 10 m (approximately 32 ft) transect, and the species ranked sixth rarest out of 38 coral species (Alcolado et al. 2010). In a study of pillar coral demographics at Providencia Island, Colombia, a total of 283 pillar coral colonies were detected in a survey of 1.66 km^2 (0.6 mi^2) for an overall density of approximately 0.000017 colonies per 10 m^2 (approximately 100 ft^2; Acosta and Acevedo 2006). In Puerto Rico, density of pillar coral ranged from 0.003 to 0.01 colonies per m^2 with an average density of 0.03 colonies per m^2; it occurred in one to 18 percent of the sites surveyed between 2008 and 2018 (NOAA NCRMP). In the USVI, average density of pillar coral ranged between 0.0003 and 0.005 colonies per m^2 (approximately 100 ft^2); it occurred in one to six percent of the sites surveyed between 2002 and 2017 (NOAA NCRMP).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the USVI in 2017. At 153 survey locations in Puerto Rico, approximately 46 to 77 percent of pillar corals were impacted (NOAA 2018b). In a post-hurricane survey of 57 sites in Florida, no pillar coral colonies were encountered, potentially reflecting their much reduced population from disease (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the USVI, although qualitative observations indicate that damage was widespread but variable by site.

Benthic cover is generally less than 1 percent in monitoring studies. Pillar coral's average cover was 0.002 percent on patch reefs and 0.303 percent in shallow offshore reefs in annual surveys of 37 sites in the Florida Keys between 1996 and 2003 (Somerfield et al. 2008). In surveys conducted in Florida between 1996 and 2016, cover of pillar coral ranged from 0 to 0.5 percent with an average of 0.0002 percent (NOAA NCRMP). In Puerto Rico, cover of pillar coral ranged between 0 and 4 percent with an average of 0.02 percent in surveys conducted between 2001 and 2016 (NOAA NCRMP). In Dominica, pillar coral comprised less than 0.9 percent cover and was present at 13.3 percent of 31 surveyed sites (Steiner 2003). Pillar coral was observed on 1 of 7 fringing reefs surveyed off Barbados, and cover was 2.7 ± 1.4 percent (Tomascik and Sander 1987).

Other than the declining population in Florida, there are two reports of population trends from the Caribbean. In monitored photo-stations in Roatan, Honduras, cover of pillar coral increased slightly from 1.35 percent in 1996 to 1.67 percent in 1999 and then declined to 0.44 percent in 2003 and to 0.43 percent in 2005 (Riegl et al. 2009).

Pillar coral is currently uncommon to rare throughout Florida and the Caribbean. Low abundance and infrequent encounter rate in monitoring programs result in small samples sizes. The low coral cover of this species renders monitoring data difficult to extrapolate to realize trends. The few studies that report pillar coral population trends indicate a general decline at some specific sites, though it is likely that the population remains stable at other sites. Low density and gonochoric broadcast spawning reproductive mode, coupled with no observed sexual recruitment, indicate that natural recovery potential from mortality is low.

Status

Pillar coral survival is susceptible to a number of threats, but there is little evidence of population declines thus far. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because pillar coral is limited to an area with high, localized human impacts and predicted increasing threats. Pillar coral inhabits most reef environments in water depths ranging from one to twenty-five m, but is naturally rare. Estimates of absolute abundance are at least tens of thousands of colonies in the Florida Keys, and absolute abundance is higher than estimates from this location due to the occurrence of the species in many other areas throughout its range. It is a gonochoric broadcast spawner with observed low sexual recruitment. Its low abundance, combined with its geographic location, exacerbates vulnerability to extinction. This is because increasingly severe conditions within the species' range are likely to affect a high proportion of its population at any given point in time. In addition, low sexual recruitment is likely to inhibit recovery potential from mortality events, further exacerbating its vulnerability to extinction. We anticipate that pillar coral is likely to decrease in abundance in the future with increasing threats.

Critical Habitat

No critical habitat has been designated for pillar coral.

Recovery Goals

No final recovery plans currently exist for pillar coral; however, a recovery outline was published in 2014. The following short and long-term recovery goals are listed in the document:

Short-Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Decrease locally manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *D. cylindrus*, *M. ferox*, and *Orbicella* spp.

Long-Term Goals:

- Cultivate and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

6.2.5.4 Rough Cactus Coral (Mycetophyllia ferox)

On September 10, 2014, NMFS listed rough cactus coral as threatened (79 FR 53851).

Species Description and Life History

Rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea (Figure 12).

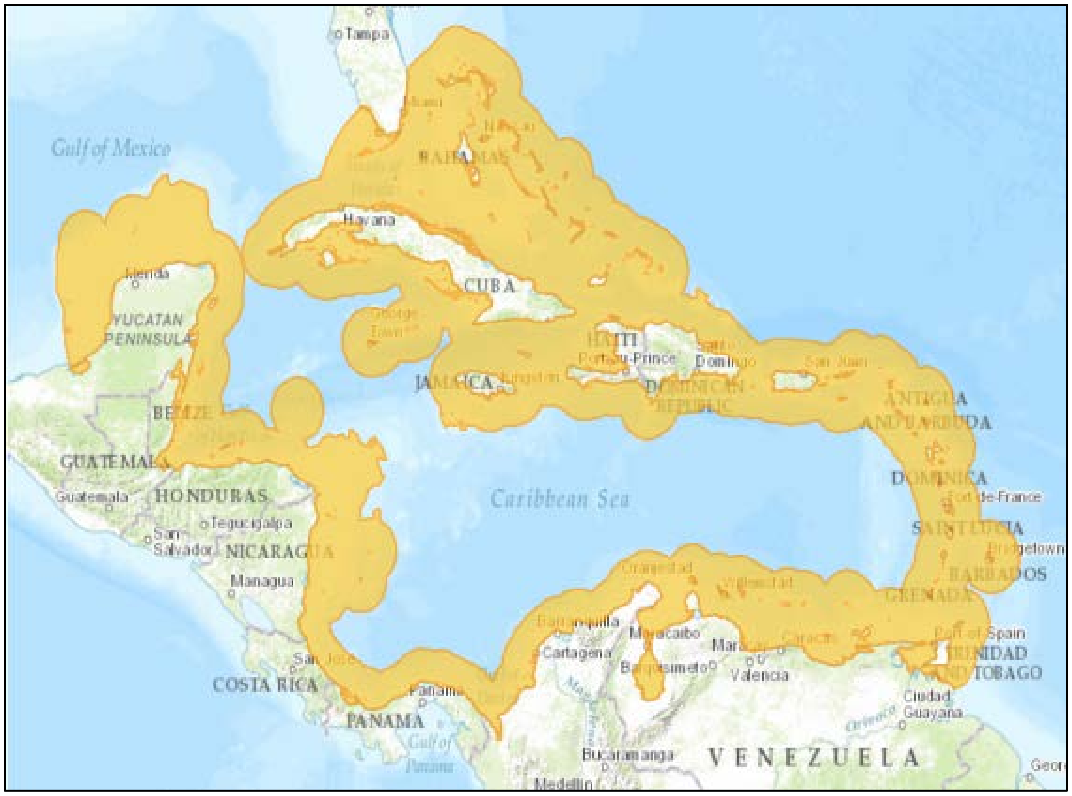


Figure 12. Range map for rough cactus coral (from Aronson et al. 2008e)

Rough cactus coral forms a thin, encrusting plate that is weakly attached to substrate. Rough cactus coral is taxonomically distinct (i.e., separate species), though difficult to distinguish in the field from other *Mycetophyllia* species.

According to the IUCN Species Account and the CITES species database, rough cactus coral occurs throughout the U.S. waters of the western Atlantic but has not been reported from Flower Garden Banks (Hickerson et al. 2008) or in Bermuda. The following areas include locations within federally protected waters where rough cactus coral has been observed and recorded (cited in Brainard et al. 2011): Dry Tortugas National Park; Virgin Island National Park/Monument; Florida Keys National Marine Sanctuary; Navassa Island National Wildlife Refuge; Biscayne National Park; Buck Island Reef National Monument. It inhabits reef environments in water depths of five to ninety m, including shallow and mesophotic habitats (e.g., > 30 m).

Rough cactus coral is a hermaphroditic brooding species. Colony size at first reproduction is greater than 100 cm^2. Recruitment of rough cactus coral appears to be very low, even in studies from the 1970s. Rough cactus coral has a lower fecundity compared to other species in its genus (Morales Tirado 2006). Over a ten-year period, no colonies of rough cactus coral were observed to recruit to an anchor-damaged site in the U.S. Virgin Islands, although adults were observed on the adjacent reef (Rogers and Garrison 2001). No other life history information appears to exist for rough cactus coral.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the rough cactus coral.

Rough cactus coral is usually uncommon or rare according to published and unpublished records. In benthic surveys conducted in the USVI between 2002 and 2018, rough cactus corals were encountered in less than half of the survey years, and density was less than or equal to 0.001 colonies per m^2 at the one to two percent of sites where they occurred (NOAA, unpublished data). Rough cactus corals were present at eight percent of sites surveyed in Puerto Rico in 2008, but in surveys conducted between 2010 and 2018, they were found at one to four percent of surveyed sites at an average density of <0.001 to 0.004 colonies per m^2 (NOAA NCRMP). Rough cactus corals were encountered in two to 10 percent of sites surveyed in Florida between 1999 and 2006, but in surveys between 2007 and 2017, they were only encountered in three survey years and at only one percent of sites at an average density of <0.001 colonies per m^2 (NOAA, unpublished data). Density of rough cactus coral in southeast Florida and the Florida Keys was approximately 0.8 colonies per approximately 100 ft^2 (10 m^2) between 2005 and 2007 (Wagner et al. 2010). In a survey of 97 stations in the Florida Keys, rough cactus coral declined in occurrence from 20 stations in 1996 to four stations in 2009 (Brainard et al. 2011). At 21 stations in the Dry Tortugas, rough cactus coral declined in occurrence from eight stations in 2004 to three stations in 2009 (Brainard et al. 2011). Taken together, these data indicate that the species has declined in Florida and potentially also in Puerto Rico over the past one to two decades.

A recent coral disease event has greatly affected coral populations in Florida. This unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species, including *Mycetophyllia* species. At study sites in southeast Florida, prevalence of disease was recorded in 67 percent of all coral colonies and 81percent of colonies of those species susceptible to the disease (Precht et al. 2016). No species-specific information is available for the effects of disease on rough cactus coral, but in a survey of 134 sites conducted between October 2017 and April 2018, nine percent of *Mycetophyllia* species were affected (Neely 2018). This disease prevalence is a snapshot in time and does not represent the total proportion of *Mycetophyllia* species affected by the disease outbreak.

Average benthic cover of rough cactus coral in the Red Hind Marine Conservation District off St. Thomas, USVI, which includes mesophotic coral reefs, was 0.003 percent in 2007, accounting for 0.02 percent of coral cover, and ranking 19 out of 21 coral species (Nemeth et al. 2008; Smith et al. 2010). In the USVI between 2001 and 2012, rough cactus coral appeared in 12 of 33 survey sites and accounted for 0.01 percent of the colonized bottom and 0.07 percent of the coral cover, ranking as 13^{th} most common coral on the reef (Smith 2013).

In other areas of the Caribbean, rough cactus coral is also uncommon. In a survey of Utila, Honduras between 1999 and 2000, rough cactus coral was observed at eight percent of 784 surveyed sites and was the 36th most commonly observed out of 46 coral species; other *Mycetophyllia* species were seen more commonly (Afzal et al. 2001). In surveys of remote southwest reefs of Cuba, rough cactus coral was observed at one of 38 reef-front sites, where average abundance was 0.004 colonies per approximately 108 ft^2 (10 m^2); this was comparatively lower than the other three *Mycetophyllia* species observed (Alcolado et al. 2010). Between 1998 and 2004, rough cactus coral was observed at three of six sites monitored in Colombia, where their cover ranged from 0.3 to 0.4 percent (Rodriguez-Ramirez et al. 2010).

Rough cactus coral has been reported to occur on a low percentage of surveyed reefs and is one of the least common coral species observed. On reefs where rough cactus coral is found, it generally occurs at abundances of less than one colony per approximately 100 ft^2 (10 m^2) and cover of less than 0.1 percent. Low encounter rate and percent cover coupled with the tendency to include *Mycetophyllia* spp. at the genus level make it difficult to discern population trends of rough cactus coral from monitoring data. However, reported losses of rough cactus coral from monitoring stations in the Florida Keys and Dry Tortugas (63-80 percent loss) and decreased encounter frequency in Puerto Rico indicate the population has declined. Based on declines in Florida and assumed declines elsewhere, we conclude rough cactus coral has likely declined throughout its range and will continue to decline based on increasing threats. As a result, it is presumed that genetic diversity for the species is low.

Status

Rough cactus coral has declined due to disease in at least a portion of its range and has low recruitment, which limits its capacity for recovery from mortality events and exacerbates vulnerability to extinction. Its depth range of 5 to 90 m moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters. Acidification is predicted to accelerate most in deeper and cooler waters than those in which the species occurs. Its habitat includes shallow and mesophotic reefs which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Rough cactus coral is usually uncommon to rare throughout its range. Its abundance, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time.

Critical Habitat

No critical habitat has been designated for rough cactus coral.

Recovery Goals

No final recovery plan currently exists for rough cactus coral, however a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved for the five coral species listed in September 2014. The recovery goals are the same for all five species (see Section 7.2.6.3) with short and long-term goals.

6.2.5.5 Lobed Star, Mountainous Star, and Boulder Star Coral (Orbicella annularis, Orbicella faveolata, and Orbicella franksi)

On September 10, 2014, NMFS listed lobed star, mountainous star, and boulder star coral as threatened (79 FR 53851).

Species Description

Lobed, mountainous, and boulder star coral occur in the western Atlantic and greater Caribbean as well as the Flower Garden Banks. Lobed and mountainous star coral may be absent from Bermuda (Figure 13).

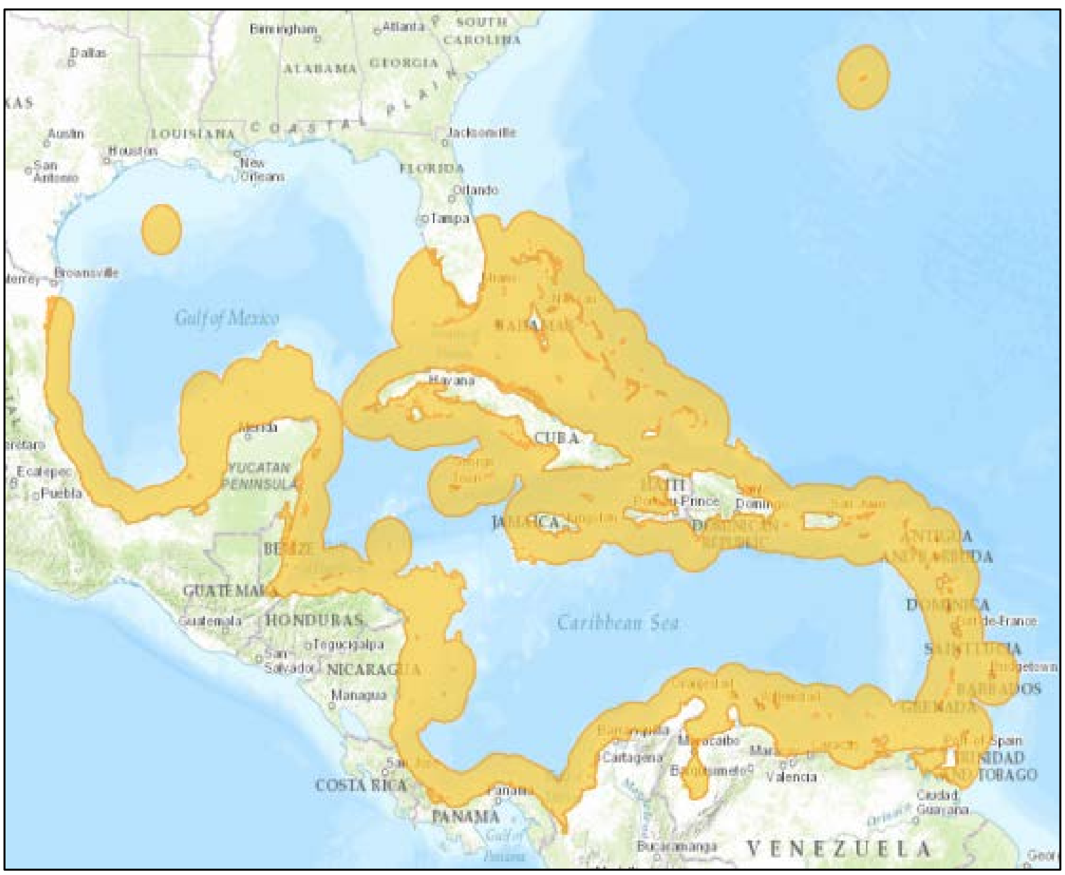


Figure 13. Range map for lobed, mountainous, and boulder star corals. Note that only boulder star corals are reported in the Bahamas (from Aronson et al. 2008b;c;d)

Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the three species in the *Orbicella annularis* star coral complex. These three species were formerly in the genus *Montastraea*; however, recent work has reclassified the three species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species – *Montastraea annularis* – with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata* and *Montastraea* (now *Orbicella*) *franksi*. The three species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into three species.

Some studies report on the star coral species complex rather than individual species since visual distinction can be difficult where colony morphology cannot be discerned (e.g. small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. However, information about *Orbicella annularis* published prior to 1994 will be attributed to the species complex since it is dated prior to the split of *Orbicella annularis* into three separate species.

Lobed Star Coral

Lobed star coral colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other two star coral species, margins on the sides of columns are typically dead. Live colony surfaces usually lack ridges or bumps.

Lobed star coral is reported from most reef environments within the Caribbean (except for Bermuda) in depths of approximately 0.5-20 m. The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >30 m) reefs, suggesting the potential for deep refuge across a broader depth range, but lobed star coral is generally described with a shallower distribution.

Mountainous Star Coral

Mountainous star coral grows in heads or sheets, the surface of which may be smooth or have keels or bumps. The skeleton is much less dense than in the other two star coral species. Colony diameters can reach up to 33 ft (10 m) with heights of 13-16 ft (4-5 m).

Mountainous star coral occurs in the western Atlantic and throughout the Caribbean, including Bahamas, Flower Garden Banks, and the entire Caribbean coastline. There is conflicting information on whether or not it occurs in Bermuda. Mountainous star coral has been reported in most reef habitats and is often the most abundant coral at 33-66 ft (10-20 m) in fore-reef environments. The depth range of mountainous star coral has been reported as approximately 1.5-132 ft (0.5-40 m), though the species complex has been reported to depths of 295 ft (90 m), indicating mountainous star coral's depth distribution is likely deeper than 132 ft (40 m). Star coral species are a common, often dominant component of Caribbean mesophotic reefs (e.g., > 100 ft [30 m]), suggesting the potential for deep refugia for mountainous star coral.

Boulder Star Coral

Large, unevenly arrayed polyps that give the colony its characteristic irregular surface distinguish boulder star coral. Colony form is variable, and the skeleton is dense with poorly developed annual bands. Colony diameter can reach up to 5 m with a height of up to 2 m.

Boulder star coral is distributed in the western Atlantic Ocean and throughout the Caribbean Sea including in the Bahamas, Bermuda, and the Flower Garden Banks. Boulder star coral tends to have a deeper distribution than the other two species in the *Orbicella* species complex. It occupies most reef environments and has been reported from water depths ranging from approximately 16-165 ft (5-50 m), with the species complex reported to 250 ft (90 m). *Orbicella* species are a common, often dominant, component of Caribbean mesophotic reefs (e.g., >100 ft [30 m]), suggesting the potential for deep refugia for boulder star coral.

Life history

The star coral species complex has growth rates ranging from 0.06-1.2 cm per year and averaging approximately one cm in linear growth per year. The reported growth rate of lobed star coral is 0.4 to 1.2 cm per year (Cruz-Piñón et al. 2003; Tomascik 1990). They grow slower in deep and murky waters.

All three species of the star coral complex are hermaphroditic broadcast spawners, with spawning concentrated on six to eight nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All three species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Further, mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about one to two hours earlier. Fertilization success measured in the field was generally below 15 percent for all three species, as it is closely linked to the number of colonies concurrently spawning. Lobed star coral is reported to have slightly smaller egg size and potentially smaller size/age at first reproduction that the other two species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 83 cm^2.

Successful recruitment by the star coral complex species has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 12 m^2 of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

In addition to low recruitment rates, species in the star coral complex have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies.

Lobed Star Coral

The reported growth rate of lobed star coral is 0.4 to 1.2 cm per year (Cruz-Piñón et al. 2003; Tomascik 1990). Asexual fission and partial mortality can lead to multiple clones of the same colony. The percentage of unique individuals is variable by location and is reported to range between 18 and 86 percent (thus, 14-82 percent are clones). Colonies in areas with higher disturbance from hurricanes tend to have more clonality. Genetic data indicate that there is some population structure in the eastern, central, and western Caribbean with population connectivity within but not across areas. Although lobed star coral is still abundant, it may exhibit high clonality in some locations, meaning that there may be low genetic diversity.

Mountainous Star Coral

Life history characteristics of mountainous star coral is considered intermediate between lobed star coral and boulder star coral especially regarding growth rates, tissue regeneration, and egg size. Spatial distribution may affect fecundity on the reef, with deeper colonies of mountainous star coral being less fecund due to greater polyp spacing. Reported growth rates of mountainous star coral range between 0.12 and 0.64 in (0.3 and 1.6 cm) per year (Cruz-Piñón et al. 2003; Tomascik 1990; Villinski 2003; Waddell 2005). Graham and van Woesik (2013) report that 44 percent of small colonies of mountainous star coral in Puerto Morelos, Mexico that resulted from partial colony mortality produced eggs at sizes smaller than those typically characterized as being mature. The number of eggs produced per unit area of smaller fragments was significantly less than in larger size classes. Szmant and Miller (2005) reported low post-settlement survivorship for mountainous star coral transplanted to the field with only 3-15 percent remaining alive after 30 days. Post-settlement survivorship was much lower than the 29 percent observed for elkhorn coral after 7 months (Szmant and Miller 2005).

Boulder Star Coral

Of 351 boulder star coral colonies observed to spawn at a site off Bocas del Toro, Panama, 324 were unique genotypes. Over 90 percent of boulder star coral colonies on this reef were the product of sexual reproduction, and 19 genetic individuals had asexually propagated colonies made up of 2 to 4 spatially adjacent clones of each. Individuals within a genotype spawned more synchronously than individuals of different genotypes. Additionally, within 16 ft (5 m), colonies nearby spawned more synchronously than farther spaced colonies, regardless of genotype. At distances greater than 16 ft (5 m), spawning was random between colonies (Levitan et al. 2011).

Population Dynamics

Lobed Star Coral

Information on lobed star coral status and populations dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Lobed star coral has been described as common overall. Demographic data collected in Puerto Rico over 9 years before and after the 2005 bleaching event showed that population growth rates were stable in the pre-bleaching period (2001–2005) but declined one year after the bleaching event. Population growth rates declined even further 2 years after the bleaching event, but they returned and then stabilized at the lower rate the following year.

In the Florida Keys, abundance of lobed star coral ranked 30 out of 47 coral species in 2005, 13 out of 43 in 2009, and 12 out of 40 in 2012. Extrapolated population estimates from stratified random samples were 5.6 million ± 2.7 million (SE) in 2005, 11.5 million ± 4.5 million (SE) in 2009, and 24.3 million ± 12.4 million (SE) in 2012. Size class distribution was somewhat variable between survey years, with a larger proportion of colonies in the smaller size classes in 2005 compared to 2009 and 2012 and a greater proportion of colonies in the greater than 36-in (90 cm) size class in 2012 compared to 2005 and 2009. Partial colony mortality was lowest at less than 4 in (10 cm; as low as approximately 5 percent) and up to approximately 70 percent in the larger size classes. In the Dry Tortugas, Florida, abundance of lobed star coral ranked 41 out of 43 in 2006 and 31 out of 40 in 2008. The extrapolated population estimate was 0.5 million ± 0.3 million (SE) colonies in 2008. Differences in population estimates between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

Colony density varies by habitat and location, and ranges from less than 0.1 to greater than one colony per approximately 100 ft^2 (10 m^2). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of lobed star coral ranged between 0.09 and 0.84 colonies per approximately 100 ft^2 (10 m^2) and was highest on mid-channel reefs followed by inshore reefs, offshore patch reefs, and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami (0.34 colonies per approximately 100 ft^2 [10 m^2]) compared to Palm Beach and Broward Counties (0.04 colonies per ~100 ft2 [10 m^2]; Burman et al. 2012). In surveys between 2005 and 2007 along the Florida reef tract from Martin County to the lower Florida Keys, density of lobed star coral was approximately 1.3 colonies per approximately 100 ft^2 (10 m^2; Wagner et al. 2010). Off southwest Cuba on remote reefs, lobed star coral density was 0.31 ± 0.46 (SE) per approximately 30 ft (10 m) transect on 38 reef-crest sites and 1.58 ± 1.29 colonies per approximately 30 ft (10 m) transect on 30 reef-front sites. Colonies with partial mortality were far more frequent than those with no partial mortality which only occurred in the size class less than 40 in (100 cm; Alcolado et al. 2010).

Recent events have greatly impacted coral populations in Florida and the US Caribbean. An unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. Lobed star coral was one of the species in surveys that showed the highest prevalence of disease, and populations were reduced to less than 25 percent of the initial population size (Precht et al. 2016).

At 153 survey locations in Puerto Rico, approximately 43-44 percent of lobed star corals were impacted by hurricanes Irma and Maria in 2017 (NOAA 2018a). In Florida, approximately 80 percent of lobed star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the USVI, though qualitative observations indicate that damage was widespread but variable by site.

Population trends are available from a number of studies. In a study of sites inside and outside a marine protected area (MPA) in Belize, lobed star coral cover declined significantly over a 10-year period (1998/99 to 2008/09; Huntington et al. 2011). In a study of 10 sites inside and outside of a marine reserve in the Exuma Cays, Bahamas, cover of lobed star coral increased between 2004 and 2007 inside the protected area and decreased outside the protected area (Mumby and Harborne 2010). Between 1996 and 2006, lobed star coral declined in cover by 37 percent in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008). Cover of lobed star coral declined 71 percent in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Mountainous Star Coral

Information on mountainous star coral status and populations dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Information regarding population structure is limited. Observations of mountainous star coral from 182 sample sites in the upper and lower Florida Keys and Mexico showed 3 well-defined populations based on 5 genetic markers, but the populations were not stratified by geography, indicating they were shared among the 3 regions (Baums et al. 2010). Of 10 mountainous star coral colonies observed to spawn at a site off Bocas del Toro, Panama, there were only three genotypes (Levitan et al. 2011) potentially indicating 30 percent clonality.

A multi-year disease event, which began in 2014, has had severe effects, causing mortality of millions of coral colonies across several species, including mountainous star coral. At 153 survey locations in Puerto Rico, approximately 12-14 percent of mountainous star corals were impacted by hurricanes Irma and Maria in 2017 (NOAA 2018a). In Florida, approximately 24 percent of mountainous star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data).

Extrapolated population estimates from stratified random samples in the Florida Keys were 39.7 ± 8 million (SE) colonies in 2005, 21.9 ± 7 million (SE) colonies in 2009, and 47.3 ± 14.5 million (SE) colonies in 2012. The greatest proportion of colonies tended to fall in the 4-8 in (10-20 cm) and 8-12 in (20-30 cm) size classes in all survey years, but there was a fairly large proportion of colonies in the greater than 36-in (90 cm)-size class. Partial mortality of the colonies was between 10 percent and 60 percent of the surface across all size classes. In the Dry Tortugas, Florida, mountainous star coral ranked seventh most abundant out of 43 coral species in 2006 and fifth most abundant out of 40 in 2008. Extrapolated population estimates were 36.1 ± 4.8 million (SE) colonies in 2006 and 30 ± 3.3 million (SE) colonies in 2008. The size classes with the largest proportion of colonies were 4-8 in (10-20 cm) and 8-12 in (20-30 cm), but there was a large proportion of colonies in the greater-than-36-in (90 cm) size class. Partial mortality of the colonies ranged between approximately 2 percent and 50 percent. Because these population abundance estimates are based on random surveys, differences between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80 percent of the sites at 1-10 percent cover (Steiner 2003). In a 1995 survey of 16 reefs in the Florida Keys, mountainous star coral ranked as the coral species with the second highest percent cover (Murdoch and Aronson 1999). On 84 patch reefs (10 ft [3 m] to 16.5 ft [5 m] depth) spanning 149 mi (240 km) in the Florida Keys, mountainous star coral was the third most abundant coral species comprising 7 percent of the 17,568 colonies encountered. It was present at 95 percent of surveyed reefs between 2001 and 2003 (Lirman and Fong 2007). In surveys of 280 sites in the upper Florida Keys in 2011, mountainous star coral was present at 87 percent of sites visited (Miller et al. 2011). In 2003 on the East Flower Garden Bank, mountainous star coral comprised 10 percent of the 76.5 percent coral cover on reefs 105-132 ft (32-40 m), and partial mortality due to bleaching, disease, and predation were rare at monitoring stations (Precht et al. 2005).

Colony density ranges from approximately 0.1-1.8 colonies per 108 ft^2 (10 m^2) and varies by habitat and location. In surveys along the Florida reef tract from Martin County to the lower Florida Keys, density of mountainous star coral was approximately 1.6 colonies per 108 ft^2 (10 m^2; Wagner et al. 2010). On remote reefs off southwest Cuba, density of mountainous star coral was 0.12 ± 0.20 (SE) colonies per 33 ft (10 m) transect on 38 reef-crest sites and 1.26 ± 1.06 (SE) colonies per 33 ft (10 m) transect on 30 reef-front sites (Alcolado et al. 2010). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of mountainous star coral ranged between 0.17 and 1.75 colonies per 108 ft^2 (10 m^2) and was highest on mid-channel reefs followed by offshore patch reefs and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami at 0.94 colonies per 108 ft^2 (10 m^2) compared to 0.11 colonies per 108 ft^2 (10 m^2) in Palm Beach and Broward Counties (Burman et al. 2012).

Boulder Star Coral

Information on boulder star coral status and population dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Reported density is variable by location and habitat and is reported to range from 0.002 to 10.5 colonies per approximately 100 ft^2 (10 m^2). Benthic surveys conducted in Florida between 1999 and 2017 recorded an average density of 0.01 to 0.36 colonies per m^2 and boulder star coral was observed at five to 45 percent of surveyed sites (NOAA, unpublished data). In Puerto Rico, boulder star coral was observed at three to 50 percent of sites, and average density ranged from 0.002 to 0.13 colonies per m^2 in surveys conducted between 2008 and 2018 (NOAA NCRMP). In the USVI, boulder star coral was present at a density of 0.02 to 0.24 colonies per m^2 in 19 to 69 percent of sites surveyed between 1999 and 2018 (NOAA, unpublished data). Limited surveys in the Flower Garden Banks reported a relatively stable density of 0.91 to 1.05 colonies per m^2 between 2010 and 2015, and boulder star coral was present at 90 to 100 percent of surveyed sites (NOAA NCRMP). In a survey of 31 sites in Dominica between 1999 and 2002, boulder star coral was present in seven percent of the sites at less than one percent cover (Steiner 2003). On remote reefs off southwest Cuba, colony density was 0.08 colonies per ~100 ft^2 (10 m^2) at 38 reef-crest sites and 1.05 colonies per ~100 ft^2 (10 m^2) at 30 reef-front sites (Alcolado et al. 2010). The number of boulder star coral colonies in Cuba with partial colony mortality were far more frequent than those with no mortality across all size classes, except for one (i.e., less than approximately 20 in [50 cm]) that had similar frequency of colonies with and without partial mortality (Alcolado et al. 2010).

Abundance at some sites in Curaçao and Puerto Rico appeared to be stable over an 8-10 year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at nine sites off Mona and Desecheo Islands, Puerto Rico. In 1998, four percent of all corals at six sites surveyed off Mona Island were boulder star coral colonies, and approximately five percent were boulder star corals in 2008; at Desecheo Island, about two percent of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

The multi-year disease event that began in 2014 caused mortality of millions of coral colonies across several species, including boulder star coral. At 153 survey locations in Puerto Rico, approximately 10-14 percent of boulder star corals were impacted by hurricanes Irma and Maria in 2017 (NOAA 2018a). In Florida, approximately 23 percent of boulder star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data).

The star coral species complex has growth rates ranging from 0.06-1.2 cm per year and averaging approximately one-cm linear growth per year. Boulder star coral is reported to be the slowest of the three species in the complex (Brainard et al. 2011). They grow slower in deep or murky waters.

Of 351 boulder star coral colonies observed to spawn at a site off Bocas del Toro, Panama, 324 were unique genotypes. Over 90 percent of boulder star coral colonies on this reef were the product of sexual reproduction, and 19 genetic individuals had asexually propagated colonies made up of two to four spatially adjacent clones of each. Individuals within a genotype spawned more synchronously than individuals of different genotypes. Additionally, within five m, colonies nearby spawned more synchronously than farther spaced colonies, regardless of genotype. At distances greater than five m, spawning was random between colonies (Levitan et al. 2011).

Status

<u>Lobed star coral</u>

Lobed star coral was historically considered one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined to between 37 percent and 90 percent over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, USVI, and St. Kitts and Nevis. Based on population estimates, there are at least tens of millions of lobed star coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. Lobed star coral remains common in occurrence. Abundance has decreased in some areas to between 19 percent and 57 percent and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, USVI, and St. Kitts and Nevis. At some reefs, a large proportion of the population is comprised of non-fertile or less-reproductive size classes. Several population projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. We conclude that while substantial population decline has occurred in lobed star coral, it is still common throughout the Caribbean and remains one of the dominant species numbering at least in the tens of millions of colonies. We conclude that the buffering capacity of lobed star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

In the Florida Keys, abundance of lobed star coral ranked 30 out of 47 coral species in 2005, 13 out of 43 in 2009, and 12 out of 40 in 2012. Extrapolated population estimates from stratified random samples were 5.6 million ± 2.7 million (SE) in 2005, 11.5 million ± 4.5 million (SE) in 2009, and 24.3 million ± 12.4 million (SE) in 2012. Size class distribution was somewhat variable between survey years, with a larger proportion of colonies in the smaller size classes in

2005 compared to 2009 and 2012 and a greater proportion of colonies in the greater than 90 cm size class in 2012 compared to 2005 and 2009. Partial colony mortality was lowest at less than ten cm (as low as approximately five percent) and up to approximately 70 percent in the larger size classes. In the Dry Tortugas, Florida, abundance of lobed star coral ranked 41 out of 43 in 2006 and 31 out of 40 in 2008. The extrapolated population estimate was 0.5 million ± 0.3 million (SE) colonies in 2008. Differences in population estimates between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

As noted previously, in a study of sites inside and outside a MPA in Belize, lobed star coral cover declined significantly over a ten year period (1998/99 to 2008/09; Huntington et al. 2011). In a study of ten sites inside and outside of a marine reserve in the Exuma Cays, Bahamas, cover of lobed star coral increased between 2004 and 2007 inside the protected area and decreased outside the protected area (Mumby and Harborne 2010). Between 1996 and 2006, lobed star coral declined in cover by 37 percent in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008). Cover of lobed star coral declined 71 percent in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Asexual fission and partial mortality can lead to multiple clones of the same colony. The percentage of unique individuals is variable by location and is reported to range between 18 percent and 86 percent (thus, 14-82 percent are clones). Colonies in areas with higher disturbance from hurricanes tend to have more clonality. Genetic data indicate that there is some population structure in the eastern, central, and western Caribbean with population connectivity within but not across areas. Although lobed star coral is still abundant, it may exhibit high clonality in some locations, meaning that there may be low genetic diversity.

Lobed star coral has undergone major declines mostly due to warming-induced bleaching and disease. Several population projections indicate population decline in the future is likely at specific sites and that local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and increased disease severity with nutrient enrichment. Lobed star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate vulnerability to extinction. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes, as has been observed in locations in the species' range. Its absolute population abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined and is higher than the estimate from these two locations due to the occurrence of the species in many other areas throughout its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because lobed star coral is limited to an area with highly localized human impacts and predicted increasing threats. Star coral occurs in most reef habitats 0.5-20 m in depth which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience high temperature variation and ocean chemistry at any given point in time. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Mountainous Star Coral

Population trend data exists for several locations. At nine sites off Mona and Desecheo Islands, Puerto Rico, no species extirpations were noted at any site over 10 years of monitoring between 1998 and 2008 (Bruckner and Hill 2009). Both mountainous star coral and lobed star coral sustained large losses during the period. The number of colonies of mountainous star coral decreased by 36 percent and 48 percent at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27 percent of all corals at six sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11 percent in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12 percent of all coral colonies were mountainous star coral in 2000, compared to seven percent in 2008.

In a survey of 185 sites in five countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) between 2010 and 2011, size of mountainous star coral colonies was significantly greater than boulder star coral and lobed star coral. The total mean partial mortality of mountainous star coral at all sites was 38 percent. The total live area occupied by mountainous star coral declined by a mean of 65 percent, and mean colony size declined from 43 ft^2 to 15 ft^2 (4005 cm^2 to 1413 cm^2). At the same time, there was a 168 percent increase in small tissue remnants less than five ft^2 (500 cm^2), while the proportion of completely live large (1.6 ft^2 to 32 ft^2 [1,500- 30,000 cm^2]) colonies decreased. Mountainous star coral colonies in Puerto Rico were much larger and sustained higher levels of mortality compared to the other four countries. Colonies in Bonaire were also large, but they experienced much lower levels of mortality. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012).

Based on population estimates, there are at least tens of millions of colonies present in each of several locations including the Florida Keys, Dry Tortugas, and the USVI. Absolute abundance is higher than the estimate from these three locations given the presence of this species in many other locations throughout its range. Population decline has occurred over the past few decades with a 65 percent loss in mountainous star coral cover across five countries. Losses of mountainous star coral from Mona and Descheo Islands, Puerto Rico include a 36-48 percent reduction in abundance and a decrease of 42-59 percent in its relative abundance (i.e., proportion relative to all coral colonies). High partial mortality of colonies has led to smaller colony sizes and a decrease of larger colonies in some locations such as The Bahamas, Bonaire, Puerto Rico, Cayman Islands, and St. Kitts and Nevis. We conclude that mountainous star coral has declined and that the buffering capacity of mountainous star coral's life history strategy, which has allowed it to remain abundant, has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

Boulder Star Coral

Information on boulder star coral status and population dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Boulder star coral is reported as common. In a 1995 survey of 16 reefs in the Florida Keys, boulder star coral had the highest percent cover of all species (Murdoch and Aronson 1999). In surveys throughout the Florida Keys, boulder star coral in 2005 ranked 26th most abundant out of 47 coral species, 32nd out of 43 in 2009, and 33rd out of 40 in 2012. Extrapolated population estimates from stratified random surveys were 8.0 ± 3.5 million (SE) colonies in 2005, 0.3 ± 0.2 million (SE) colonies in 2009, and 0.4 ± 0.4 million (SE) colonies in 2012. The authors note that differences in extrapolated abundance between years were more likely a function of sampling design rather than an indication of population trends. In 2005, the greatest proportions of colonies were in the smaller size classes of approximately 4-8 in (10-20 cm) and approximately 8-12 in (20-30 cm). Partial colony mortality ranged from 0 percent to approximately 73 percent and was generally higher in larger colonies (Miller et al. 2013).

In the Dry Tortugas, Florida, boulder star coral ranked fourth highest in abundance out of 43 coral species in 2006 and 8th out of 40 in 2008. Extrapolated population estimates were 79 ± 19 million (SE) colonies in 2006 and 18.2 ± 4.1 million (SE) colonies in 2008. The authors note the difference in estimates between years was more likely a function of sampling design rather than population decline. In the first year of the study (2006), the greatest proportion of colonies were in the size class approximately 8-12 in (20-30 cm) with twice as many colonies as the next most numerous size class and a fair number of colonies in the largest size class of greater than 3 ft (90 cm). Partial colony mortality ranged from approximately 10-55 percent. Two years later (2008), no size class was found to dominate, and proportion of colonies in the medium-to-large size classes (approximately 24-36 in) appeared to be less than in 2006. The number of colonies in the largest size class of greater than 3 ft (90 cm) remained consistent. Partial colony mortality ranged from approximately 15-75 percent (Miller et al. 2013).

In 2003, on the east Flower Garden Bank, boulder star coral comprised 46 percent of the 76.5 percent coral cover on reefs approximately 105-131 ft (32-40 m) in depth. Partial coral mortality due to bleaching, disease and predation was rare in survey stations (Precht et al. 2005). In a survey of 31 sites in Dominica between 1999 and 2002, boulder star coral was present in 7 percent of the sites at less than 1 percent cover (Steiner 2003).

Reported density is variable by location and habitat and is reported to range from 0.02 to 1.05 colonies per approximately (~) 100 ft^2 (10 m^2). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of boulder star coral ranged between 0.04 and 0.47 colonies per ~100 ft^2 (10 m^2) and was highest on the offshore patch reef and fore-reef habitats (Burman et al. 2012). In south Florida, density was highest in areas south of Miami at 0.44 colonies per ~100 ft^2 (10 m^2) compared to 0.02 colonies per ~100 ft^2 (10 m^2) in Palm Beach and Broward Counties (Burman et al. 2012). Along the Florida reef tract from Martin County to the lower Florida Keys, density of boulder star coral was ~0.9 colonies per ~100 ft^2 (10 m^2; Wagner et al. 2010). On remote reefs off southwest Cuba, colony density was 0.083 ± 0.17 (SD) per ~100 ft^2 (10 m^2) transect on 38 reef-crest sites and 1.05 ± 1.02 colonies per ~100 ft^2 (10 m^2) transect on 30 reef-front sites (Alcolado et al. 2010). The number of boulder star coral colonies in Cuba with partial colony mortality were far more frequent than those with no mortality across all size classes, except for 1 (i.e., less than ~20 in [50 cm]) that had similar frequency of colonies with and without partial mortality (Alcolado et al. 2010).

Abundance in Curaçao and Puerto Rico appears to be stable over an 8-10 year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at nine sites off Mona and Desecheo Islands, Puerto Rico. In 1998, 4 percent of all corals at six sites surveyed off Mona Island were boulder star coral colonies and approximately 5 percent in 2008; at Desecheo Island, about 2 percent of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

Based on population estimates, there are at least tens of millions of colonies present in both the Dry Tortugas and USVI. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. The frequency and extent of partial mortality, especially in larger colonies of boulder star coral, appear to be high in some locations such as Florida and Cuba, though other locations like the Flower Garden Banks appear to have lower amounts of partial mortality. In some locations, colony size has decreased over the past several decades. Bruckner (2012) conducted a survey of 185 sites (2010 and 2011) in five countries (The Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) and reported the size of boulder star coral and lobed star coral colonies as significantly smaller than mountainous star coral. The total mean partial mortality of boulder star coral was 25 percent. Overall, the total live area occupied by boulder star coral declined by a mean of 38 percent, and mean colony size declined from 210 in^2 to 131 in^2 (1356 cm^2 to 845 cm^2). At the same time, there was a 137 percent increase in small tissue remnants, along with a decline in the proportion of large (1,500 to 30,000 cm^2), completely alive colonies. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012).

A decrease in boulder star coral percent cover by 38 percent and a shift to smaller colony size across five countries suggest that population decline has occurred in some areas; colony abundance appears to be stable in other areas. We anticipate that while population decline has occurred, boulder star coral is still common with the number of colonies at least in the tens of millions. Additionally, we conclude that the buffering capacity of boulder star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Critical Habitat

No critical habitat has been designated for lobed star coral.

Recovery Goals

No final recovery plan currently exists for lobed star, mountainous star or boulder star coral; however, a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved for the five coral species listed in September 2014. The recovery goals are the same for all five species (see Section 7.2.6.3) with short and long-term goals.

6.2.6 Status of Elkhorn and Staghorn Coral Critical Habitat

On November 26, 2008, a Final Rule designating *Acropora* critical habitat was published in the Federal Register. Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species. The feature essential to the conservation of *Acropora* species (also known as the PBF) is substrate of suitable quality and availability in water depths from the mean high water line to 30 m in order to support successful larval settlement, recruitment, and reattachment of fragments. "Substrate of suitable quality and availability" means consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing this feature have been identified in four locations within the jurisdiction of the United States: the Florida area, which comprises approximately 1,329 mi^2 (3,442 km^2) of marine habitat; the Puerto Rico area, which comprises approximately 1,383 mi^2 (3,582 km^2) of marine habitat; the St. John/St. Thomas area, which comprises approximately 121 mi^2 (313 km^2) of marine habitat; and the St. Croix area, which comprises approximately 126 mi^2 (326 km^2) of marine habitat. The total area covered by the designation is thus approximately 2,959 mi^2 (7,664 km^2).

As defined in the final rule, critical habitat does not include areas subject to the 2008 Naval Air Station key West Integrated Natural Resources Management Plan; all areas containing existing (already constructed) federally authorized or permitted man-made structures such as aids-to-navigation (ATONS), artificial reefs, boat ramps, docks, pilings, maintained channels, or marinas; or twelve federal maintained harbors and channels including Fajardo Harbor, which is within the action area for this consultation.

The PBF can be found unevenly dispersed throughout the critical habitat units, interspersed with natural areas of loose sediment, fleshy or turf macroalgae covered hard substrate. Existing federally authorized or permitted man-made structures such as artificial reefs, boat ramps, docks, pilings, channels or marinas do not provide the PBF. The proximity of this habitat to coastal areas subjects this feature to impacts from multiple activities including dredging and disposal activities, stormwater run-off, coastal and maritime construction, land development, wastewater and sewage outflow discharges, point and non-point source pollutant discharges, fishing, placement of large vessel anchorages, and installation of submerged pipelines or cables. The impacts from these activities, combined with those from natural factors (i.e., major storm events), significantly affect the quality and quantity of available substrate for these threatened species to successfully sexually and asexually reproduce.

A shift in benthic community structure from coral-dominated to algae-dominated that has been documented since the 1980s means that the settlement of larvae or attachment of fragments is often unsuccessful (Hughes and Connell 1999). Sediment accumulation on suitable substrate also impedes sexual and asexual reproductive success by preempting available substrate and smothering coral recruits.

While algae, including crustose coralline algae and fleshy macroalgae, are natural components of healthy reef ecosystems, increased algal dominance since the 1980s has impeded coral recruitment. The overexploitation of grazers through fishing has also contributed to fleshy macroalgae persistence in reef and hard bottom areas formerly dominated by corals. Impacts to water quality associated with coastal development, in particular nutrient inputs, are also thought to enhance the growth of fleshy macroalgae by providing them with nutrient sources. Fleshy macroalgae are able to colonize dead coral skeleton and other hard substrate and some are able to overgrow living corals and crustose coralline algae. Because crustose coralline algae is thought to provide chemical cues to coral larvae indicating an area is appropriate for settlement, overgrowth by macroalgae may affect coral recruitment (Steneck 1986). Several studies show that coral recruitment tends to be greater when algal biomass is low (Rogers et al. 1984; Hughes 1985; Connell et al. 1997; Edmunds et al. 2004; Birrell et al. 2005; Vermeij 2006). In addition to preempting space for coral larval settlement, many fleshy macroalgae produce secondary metabolites with generalized toxicity, which also may inhibit settlement of coral larvae (Kuffner and Paul 2004). The rate of sediment input from natural and anthropogenic sources can affect reef distribution, structure, growth, and recruitment. Sediments can accumulate on dead and living corals and exposed hard bottom, thus reducing the available substrate for larval settlement and fragment attachment.

In addition to the amount of sedimentation, the source of sediments can affect coral growth. In a study of 3 sites in Puerto Rico, Torres (2001) found that low-density coral skeleton growth was correlated with increased re-suspended sediment rates and greater percentage composition of terrigenous sediment. In sites with higher carbonate percentages and corresponding low percentages of terrigenous sediments, growth rates were higher. This suggests that re-suspension of sediments and sediment production within the reef environment does not necessarily have a negative impact on coral growth while sediments from terrestrial sources increase the probability that coral growth will decrease, possibly because terrigenous sediments do not contain minerals that corals need to grow (Torres 2001).

Long-term monitoring of sites in the USVI indicate that coral cover has declined dramatically; coral diseases have become more numerous and prevalent; macroalgal cover has increased; fish of some species are smaller, less numerous, or rare; long-spined black sea urchins are not abundant; and sedimentation rates in nearshore waters have increased from one to 2 orders of magnitude over the past 15 to 25 years (Rogers et al. 2008). Thus, changes that have affected elkhorn and staghorn coral and led to significant decreases in the numbers and cover of these species have also affected the suitability and availability of habitat.

Elkhorn and staghorn corals require hard, consolidated substrate, including attached, dead coral skeleton, devoid of turf or fleshy macroalgae for their larvae to settle. Atlantic and Gulf of Mexico Rapid Reef Assessment Program data from 1997-2004 indicate that although the historic range of both species remains intact, the number and size of colonies and percent cover by both species has declined dramatically in comparison to historic levels (Ginsburg and Lang 2003).

Long-term monitoring of marine habitats in natural reserves around Puerto Rico, begun in 1999 and now at full capacity indicates statistically significant declines in live coral cover (Garcia-Sais et al. 2008). The most pronounced declines in coral cover were observed between the 2005 and 2006 surveys, corresponding to the dramatic bleaching even that occurred because of high sea surface temperatures in 2005. Declines of up to 59 percent were measured in surveyed reefs and a proportional increase in turf algae was observed (Garcia-Sais et al. 2008). Together with bleaching-associated mortality, coral disease led to the recorded loss of 50 to 80 percent live coral cover from reefs in La Parguera, Culebra, Mona, and Desecheo, Puerto Rico, and other important reefs in the northeast and southern Caribbean between 2005 and 2011 (Weil et al. 2009; Hernández-Pacheco et al. 2011; Bruckner and Hill 2009; Croquer and Weil 2009; Bastidas et al. 2012). Thus, changes that have affected elkhorn and staghorn corals and led to significant decreases in their numbers and cover have also affected the suitability and availability of habitat for these species.

7 Environmental Baseline

The “environmental baseline” refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency’s discretion to modify are part of the environmental baseline (50 C.F.R. §402.02; 84 FR 44976 published August 27, 2019).

The environmental baseline for this Opinion includes the effects of several activities that affect the survival and recovery of fin, sei, blue, and sperm whales; green, leatherback, and hawksbill sea turtles; Nassau grouper; elkhorn and staghorn corals, rough cactus coral, pillar coral, and lobed, mountainous and boulder star corals; and the ability of designated critical habitat in the action area to support its intended conservation function for elkhorn and staghorn corals.

7.1 Status of Sperm Whales; Green, Leatherback, and Hawksbill Sea Turtles; Nassau Grouper, ESA-Listed Corals, and Elkhorn and Staghorn Coral Within the Action Area

7.1.1 Sperm Whales

Sperm whales are widely distributed in the Caribbean and are common in the deep-water passages between islands and along continental slopes (CH2M Hill 2018). Marine mammal surveys conducted around Puerto Rico report observations of cow-calf pairs and juvenile sperm whales in the vicinity of Vieques (Roden and Mullin 2000; GMI 2001). Mignucci-Giannoni et al. (2000) suggested that the waters south of Vieques may be important nursing grounds for some marine mammal species, including sperm whales, and may be part of the calving grounds for this species. A contractor reported an immature sperm whale carcass washed up on the beach of Bahia Salinas del Sur in July 2013, indicating that mother-calf pairs are present at certain times of year in deeper waters within or adjacent to the action area. There are no population estimates for sperm whales in the action area.

7.1.2 Green, Leatherback, and Hawksbill Sea Turtles

The sea turtle nesting beaches around Vieques have not changed over the approximately two decades there have been surveys of nesting done by PRDNER, USFWS, and the Navy. Based on information USFWS provided to the Navy for this consultation, USFWS has observed some small trends and shifts in nesting beach use over the past 5-8 years, some of which may be associated with storm events that change beach profiles. For example, Hurricane Maria removed much of the sand from Boca Quebrada Beach but USFWS, based on a survey in September 2019 by a USFWS biologist, reports that Tropical Storm Dorian (on August 28, 2019) deposited a large amount of sand on the beach. However, USFWS also documented some beach erosion and nest losses caused by Dorian but have not fully evaluated the beaches, particularly because much of the roads to the beaches of Vieques remain out of service since Hurricane Maria (as of September 2019).

Green sea turtle nesting activity is low in Puerto Rico when compared to other areas in the Caribbean and Atlantic. The main green sea turtle nesting sites in Puerto Rico are beaches on the northeast coast of Vieques Island and the beach on the southeast coast of Caja de Muertos Island (Jiménez-Marrero 2000). Beaches in Vieques utilized by green sea turtles, normally every other year but occasionally annually, are largely on the eastern side of the island and include Barco, Blanca, and Brava (see Figure 14) based on monitoring by PRDNER, USFWS, and the Navy. Nesting on these beaches by green sea turtles is estimated by the USFWS to be the highest amount of nesting for all of Puerto Rico. Over the past 5-7 years, USFWS has observed a shift in nesting by green sea turtles to public beaches on the east of Vieques and to beaches on the southwest of the island, primarily from Playa Grande to Punta Vaca. For 2019 (up to September 6), USFWS has observed 75 green sea turtle nests in the VNWR public areas and the DNER reserve and 65 nests in the VNWR restricted area on the east end of the island for a total of 140 nests. Green sea turtle nesting has shown an upward trend since monitoring began in 1992. The number of green hatchlings that make it out of the nest onto the beach (i.e., emergent success) is 50 percent based on data from Florida (Brost et al. 2015). If we assume there are 114 eggs per nest and 140 nests are lain annually, 15,960 eggs would produce 7,980 green hatchlings (114 times 140 times 50 percent). Green sea turtles, likely adults and juveniles, are also sighted regularly in nearshore waters of UXO 16. A compilation of data from studies beginning in the 1970s and continuing through 2000 indicate that the frequency of green sea turtle in-water sightings is usually similar to that of hawksbills with about 40-50 percent of in-water sightings being one of these two species (Bauer et al. 2008).

Leatherback sea turtle nesting activity occurs on beaches around the main island of Puerto Rico, with the highest amount of leatherback nesting taking place on beaches along the northeastern coast of the island. Leatherback nesting also occurs around offshore islands of Puerto Rico, including Vieques where a number of beaches are used by this species (see Figure 14). USFWS reports that leatherback nesting used to be higher on Matias Beach (also known as Yellow Beach) than anywhere else on Vieques and Campana Beach (formerly Purple Beach) averaged more nests per season in the past. Leatherback nesting is now largely on beaches on the west end of Vieques with the greatest amount of nesting reported from Playa Grande to Punta Vaca. For 2019 (up to September 6), USFWS has observed 52 leatherback sea turtle nests in the VNWR public areas and the DNER reserve and 17 nests in the VNWR restricted area on the east end of the island for a total of 69 nests. The emergent success of leatherback hatchlings is between 38.7 and 72 percent in the United States (Eckert and Eckert 1990; Stewart and Johnson 2006; Tucker 1988). If we assume there are 77 fertile eggs per nest and 69 nests are lain annually, 5,313 leatherback eggs would produce between 2,056 and 3,825 leatherback hatchlings (77 times 69 times emergent success of 39 and 72 percent, respectively).

The distribution of hawksbill sea turtle nesting activity in Vieques makes up a small percentage of the overall nesting activity around Puerto Rico when compared to Mona Island, where the most important hawksbill nesting areas are located. Hawksbill sea turtles nest on beaches of Vieques, often throughout the year (Figure 14). According to the USFWS, hawksbills have gradually shifted nesting activity from Boca Quebrada Beach to beaches from Playa Grade to Punta Vaca on the southwest of Vieques. For 2019 (up to September 6), USFWS has observed 46 hawksbill sea turtle nests in the VNWR public areas and the DNER reserve and 12 nests in the VNWR restricted area on the east end of the island for a total of 58 nests. Hawksbill sea turtle nesting has also increased since monitoring began in 1991 around Vieques with peak nesting on Tamarindo Sur, Fanduca Beach, Jalova Beach, and Jalovita Beach most years (around 50 nests per site based on data up to 2001) and low numbers of nests on other beaches (such as 2 nests per year at Bahía Icacos; CH2M Hill 2011). Hawksbill clutch size is approximately 140 eggs and emergent success at nesting beaches in the Caribbean is approximately 80 percent (Ditmer and Stapleton 2012). If we assume there are 140 eggs per hawksbill nest and 58 nests are lain annually, there could be 8,120 eggs resulting in 6,496 hawksbill hatchlings (140 times 58 times 80 percent emergent success). Hawksbill sea turtles, likely juveniles and adults, are also sighted regularly in nearshore waters of UXO 16.

Studies have shown that sea turtle hatchling mortality rates range from 30-60 percent as the animals leave the beach and swim toward open water and only 2.5 in 1,000 reach adulthood (Pilcher 1999; Frazer 1992). Thus, between 2,394 and 4,788 green sea turtle hatchlings; between 617 and 2,295 leatherback sea turtle hatchlings; and between 1,949 and 3,898 hawksbill sea turtle hatchlings could survive to swim toward open water in UXO 16 adjacent to beaches where nesting of green, leatherback, and/or hawksbill is reported.

In order to come up with a population estimate for green sea turtles in the action area, we divide the total number of nests reported up to September 6, 2019 (140 nests) by the number of times an adult female green sea turtle nests per season, on average (2.625), we calculate there are approximately 53 adult females. If we then assume a 1:1 sex ratio, there would be 53 adult male green sea turtles in the action area as well, for a total population of 106 adult green sea turtles. Similarly, for leatherback sea turtles, if we divide 69 nests by the average number of times a single female nests per season (4.475), we calculate there are approximately 15 adult females. If we then assume a 1:1 sex ratio, there would be 15 adult male leatherback sea turtles in the action area as well, for a total population of 30 adult leatherback sea turtles. For hawksbill sea turtles in the action area, we divide the total number of nests reported up to September 6, 2019 (58 nests) by the number of times an adult female hawksbill sea turtle nests per season, on average (2.4425), and calculate there are approximately 24 adult females. If we then assume a 1:1 sex ratio, there would be 24 adult male hawksbill sea turtles in the action area as well, for a total population of 48 adult hawksbill sea turtles. This is likely an underestimate as it only includes the nesting around Vieques for a portion of the year, furthermore, nest monitoring was complicated by storms that blocked access to beaches, and NMFS does not have recent nesting data for other portions of the action area such as the coast of Fajardo, Puerto Rico.

After a pelagic post-hatchling phase, juvenile hawksbill sea turtles (20 to 25 cm carapace length) appear to establish a home range where they often stay until they are sexually mature. Immature hawksbill and green sea turtles are known to spend years in shallow water developmental habitats in nearshore areas containing a mixture of coral reefs, colonized hard bottom, and seagrass beds where they grow to sexual maturity (Meylan 1999; Makowski et al. 2006; Renaud et al. 1995). Makowski et al. (2006) tracked six green sea turtles with straight carapace lengths of 27.9 to 48.1 cm and found that all of them exhibited overlap in core foraging areas in Palm Beach, Florida. Makowski et al. (2006) found considerable overlap between refuge and foraging sites for green sea turtles with the entire home range of each turtle concentrated over the algal-rich nearshore worm reef where immature green sea turtles were shown to eat macroalgae and sponges as the dominant components of their diet. Turtles were also found to have one to two distinct nocturnal resting sites within their home ranges that were not shared with another turtle, although foraging habitats of turtles did overlap (Makowski et al. 2006). Mean home range for green sea turtles was 2.38 km^2 with ranges from 0.69 to 5.05 km^2 with foraging activity centers measuring between 0.18 and 1.17 km^2 (Makowski et al. 2006). Other studies, such as that by Lamont and Iverson (2018) similarly found that core use areas where green sea turtles forage are smaller than their overall home range with mean core use areas of 4.2 km^2 versus 15.8 km^2 home range areas. Wershoven and Wershoven (1992) found that green sea turtles enter hard bottom habitat in Broward County nearshore waters at approximately 30 cm curved carapace length and depart when they reach a length of 60 cm. Based on their study, Wershoven and Wershoven (1992) determined that there could be five immature green sea turtles per acre.

Data from in-water sea turtle surveys at Buck Island, St. Croix, indicate that the foraging grounds for juvenile and adult hawksbill sea turtles are spatially distinct (NPS 2003;2004; Hart et al. 2013; Hart et al. 2014) based on sizes of turtles captured all of which were smaller than reproductive adults.Van Dam and Diez (1998) surveyed four sites on coral reefs and along cliff walls of Mona and Monito Islands, Puerto Rico, for a 4-year period to determine the home range of immature hawksbills measuring less than 65 cm in carapace length. Van Dam and Diez (1998) found that turtles establish their home range in a particular habitat type and do not vary this habitat preference. Similarly, Witt et al. (2010) found that habitat structure influenced site fidelity for juvenile hawksbills in the British Virgin Islands. Cuevas et al. (2007) found that juvenile hawksbills in Yucatan, Mexico, showed a difference in habitat preference during the day (octocoral cover between 20% to 40%) and night (trending toward bare substrate). Mean home range for hawksbill sea turtles was approximately 2.04 km^2 with ranges from 0.05 to 4.03 km^2 in Belize (Scales et al. 2011); 1.1 to 19 km^2 using minimum convex polygon (MCP) estimates versus 0.01 to 1.2 km^2 using kernel density estimates (KDE) in Southeast Florida (Wood et al. 2017); 0.15 to 0.55 km^2 using MCP and 5.46 km^2 mean using KDE on an inshore reef in Honduras (Berube et al. 2012); and 0.07 to 0.14 km^2 in Mona and Monito Islands, Puerto Rico (Van Dam and Diez 1998). Diez and Van Dam (2002) estimated there were between 0.11 and 0.5 immature hawksbill sea turtles per acre in habitats around Mona Island, Puerto Rico.

Using the estimate of five immature green sea turtles per acre from the Wershoven and Wershoven (1992) study and the estimate of 0.11 to 0.5 immature hawksbill sea turtles per acre from the Diez and Van Dam (2002) study, we can calculate the number of juvenile green and hawksbill sea turtles that may be present in the action area. Based on the WAA, there are 3,557 acres of seagrass habitat and 5,198 acres of coral habitat, all of which may be used by juvenile green and hawksbill sea turtles. Using this information, there could be 1,751 juvenile green sea turtles and between 963 and 4,377 juvenile hawksbill sea turtles in the action area. There may be additional seagrass and coral habitat outside UXO 16 but within the action area, including in areas around Vieques and on the main island in the area of Fajardo that will be along transit routes used to access areas where activities will take place in UXO 16 so these numbers of juveniles may be underestimates.

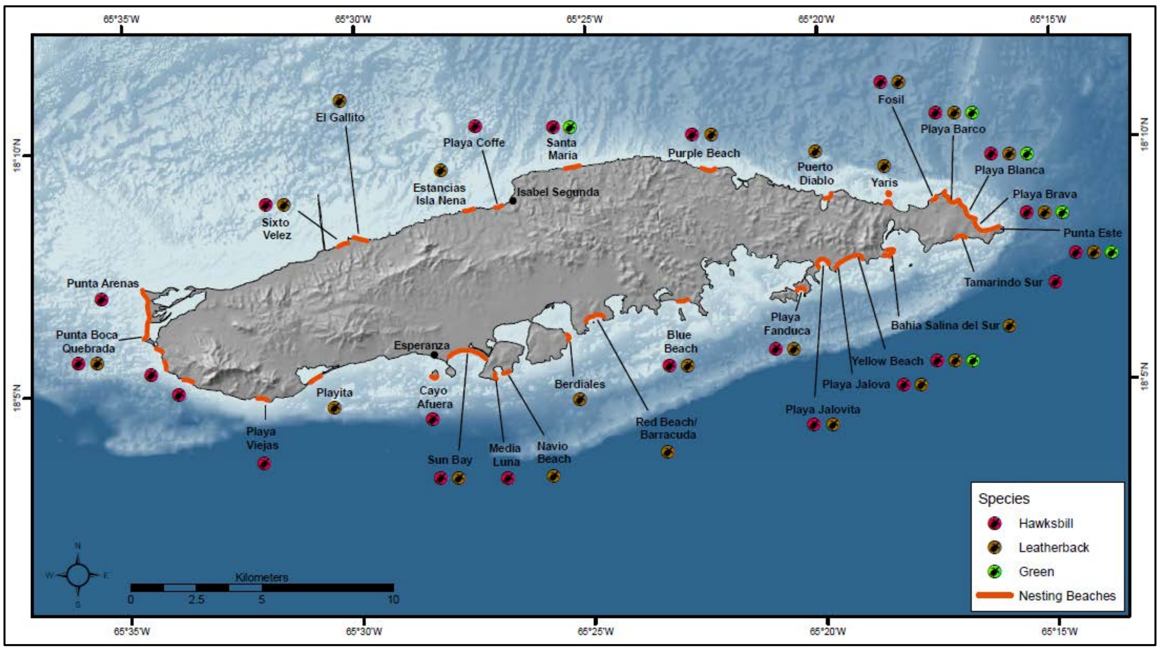


Figure 14. Sea turtle nesting beaches around Vieques compiled from data from eight studies from 1979 to 2001 (Bauer et al. 2008)

The greatest densities of sea turtles in the water are near Mosquito Pier, off the eastern (from Cayo Yallis to Bahia Salinas del Sur) and western shores, and close to Sun Bay (Figure 15; Bauer et al. 2008).

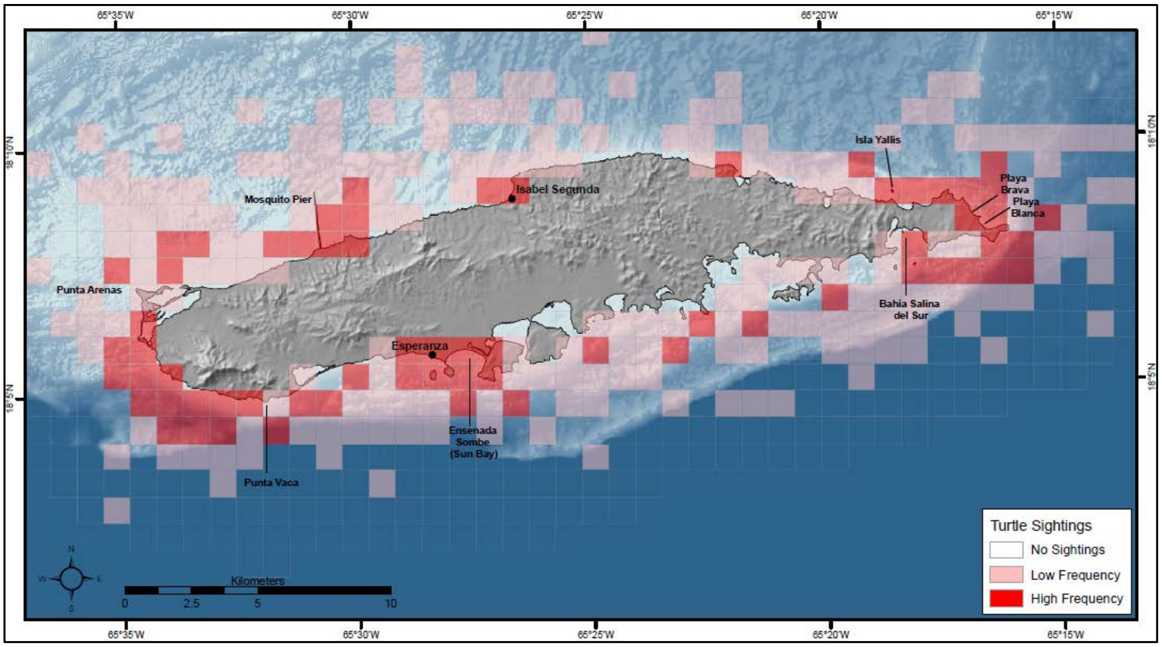


Figure 15. Compilation of in-water sea turtle sightings around Vieques from 1978 to 2001 (from Bauer et al. 2008)

7.1.3 Nassau Grouper

Juvenile Nassau grouper use nearshore seagrass beds, embayments, backreefs, and other shallower habitats while adults are common in deeper reef areas. Historic spawning aggregation sites (SPAGS) for this species are off the west coast of Puerto Rico but fishers also identified potential sites around Vieques, including the eastern point of Vieques and an area known as OP (Figure 16; Ojeda-Serrano et al. 2007). Additionally, in-water surveys conducted in waters around Vieques, including within UXO 16, in 1979, 1986, 2001, 2003, and 2004 (Figure 17) observed Nassau grouper in reef-associated habitats, including coral reefs, seagrass beds, and colonized hard bottom (Department of the Navy 1979;1986; GMI 2003; García-Sais et al. 2001; García-Sais et al. 2004). These surveys largely reported only the presence of Nassau grouper rather than the numbers of individuals of the species observed. There is no population estimate for Nassau grouper available for the action area.

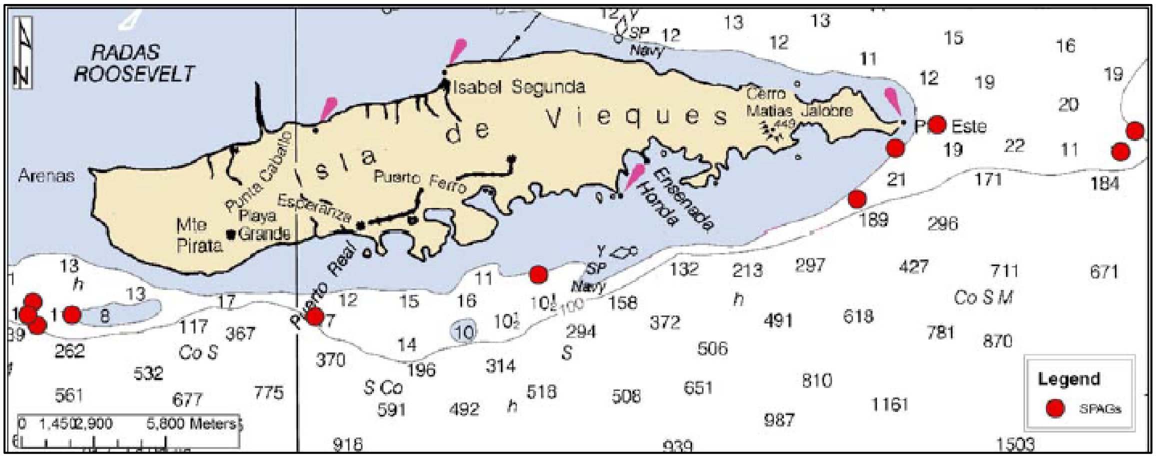


Figure 16. Potential spawning aggregation sites (SPAGs) around Vieques identified by fishers that may be used by Nassau grouper (from Ojeda-Serrano et al. 2007)

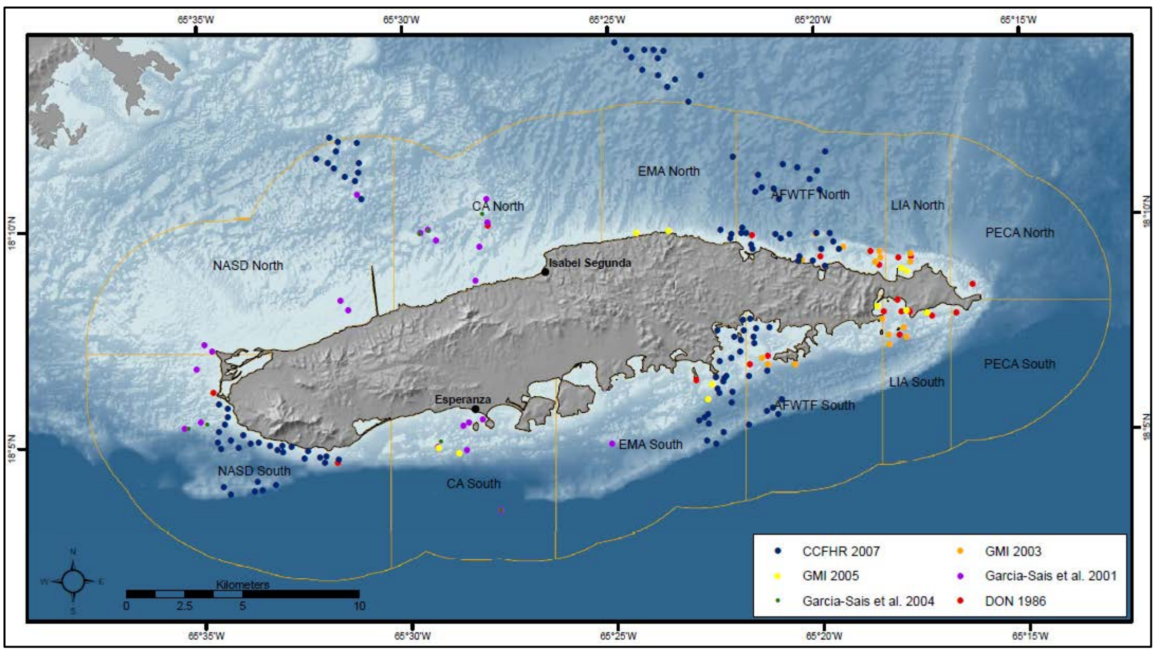


Figure 17. Location of reef fish surveys around Vieques. Surveys reporting Nassau grouper include García-Sais et al. 2001 and 2004; DON 1986; and GMI 2003 (from Bauer et al. 2008)

7.1.4 ESA-Listed Corals

There are hard bottom and reef habitats containing coral cover (Figures 18 and 19), including colonies of ESA-listed corals in waters around Vieques (Figure 20). Lobed and mountainous star corals were found to be common in surveys conducted by the Navy and its contractors within UXO 16. National Centers for Coastal Ocean Science (NCCOS) also found lobed star coral to be the dominant live coral species on reef and hard bottom habitats in sampling sites around Vieques (Bauer and Kendall 2010). Staghorn coral was observed in 12 percent of NCCOS surveys while elkhorn was observed in one percent (Bauer and Kendall 2010), but this may also be a function of the depths where surveys were conducted as elkhorn prefers depths up to 5 m.

Pillar coral, boulder star coral, mountainous star coral, and rough cactus coral have been reported in studies conducted in UXO 16 in 2001-2005 (Bauer et al. 2008). The NCRMP surveys around Puerto Rico in 2014 included sites around Vieques. Of 353 corals identified during sampling at stations around Vieques, two were lobed star corals (0.5 percent), 35 were mountainous star corals (10 percent), 70 were boulder star corals (20 percent), and 1 was pillar coral (0.28 percent). One rough cactus coral colony was also observed in a survey of two sites around Vieques targeting ESA-listed corals in deeper waters. In 2016, only three mountainous star corals and no other ESA-listed coral species were observed during NCRMP surveys around Vieques targeting ESA-listed corals, but other NCRMP benthic surveys in 2016 reported elkhorn, staghorn, pillar, rough cactus, and lobed, mountainous and boulder star coral colonies in survey sites around Vieques. Following the 2017 hurricanes, 20 sites around Vieques were surveyed to assess the condition of corals. Thirteen percent of corals surveyed around Vieques, totaling thousands of colonies, suffered some damage from the hurricanes but overall damage was minor (defined as a site with 49 or fewer broken colonies and fragments; NOAA 2018a).

Overall, coral cover in reefs and hard bottoms was found to be low. This does not appear to be due to the size of the minimum mapping unit used for the benthic map creation, which was 1,000 m^2 (0.25 acre; CH2M Hill 2018) because NCCOS found hard coral cover ranged from less than 2 percent in sampling sites of eastern Vieques to 6.7 percent in southwestern sampling sites (Bauer and Kendall 2010) in diver surveys.

CH2M Hill (2018) estimated that there could be up to 5,173 ESA-listed coral colonies affected by the activities that are part of the proposed action. This is based in part on the estimate of 5,198 acres of coral habitats from the results of the WAA. There are likely to be more ESA-listed coral colonies than this as the estimate only includes colonies likely to be adversely affected by the proposed action due to their location in relation to MEC/MPPEH. In addition, no coral survey data are detailed enough to enable a determination of the numbers of colonies of each ESA-listed coral species in the action area because none of the surveys included quantification of ESA-listed corals by species.

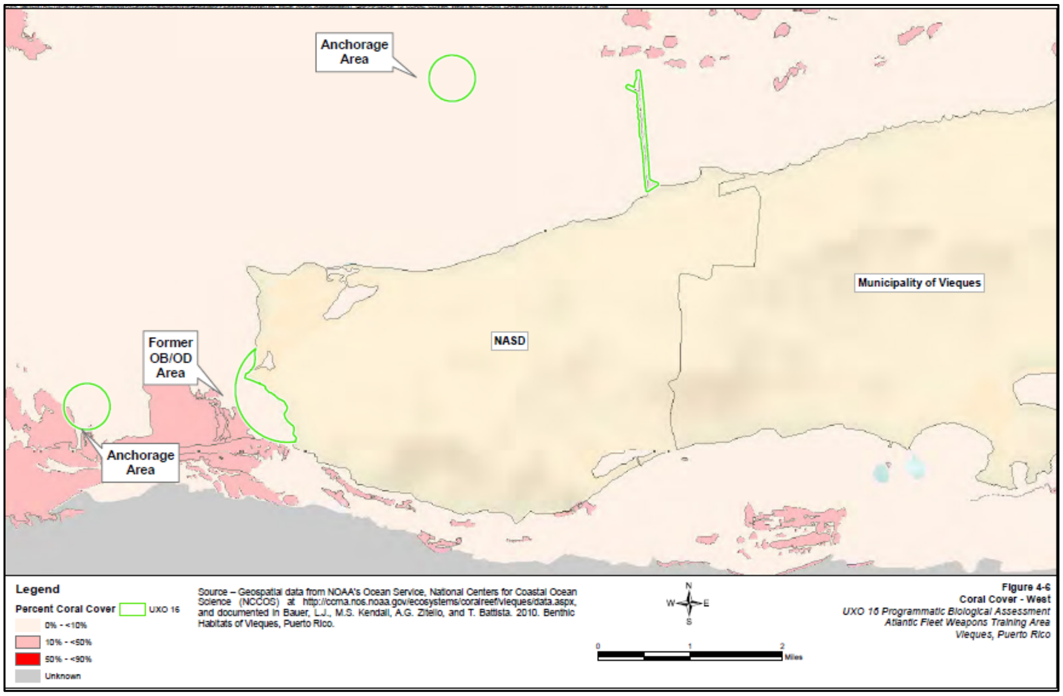


Figure 18. Estimated percent coral cover on the western side of Vieques showing areas within UXO 16 (outlined in green; from CH2M Hill 2018)

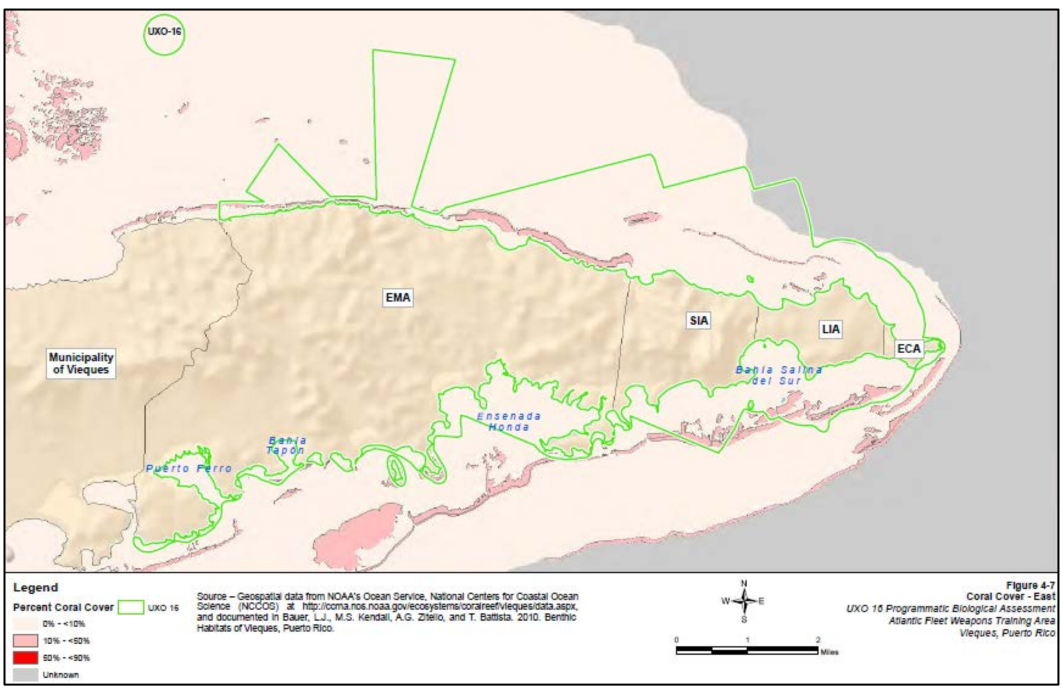


Figure 19. Estimated percent coral cover on the eastern side of Vieques showing areas within UXO 16 (outlined in green; from CH2M Hill 2018)

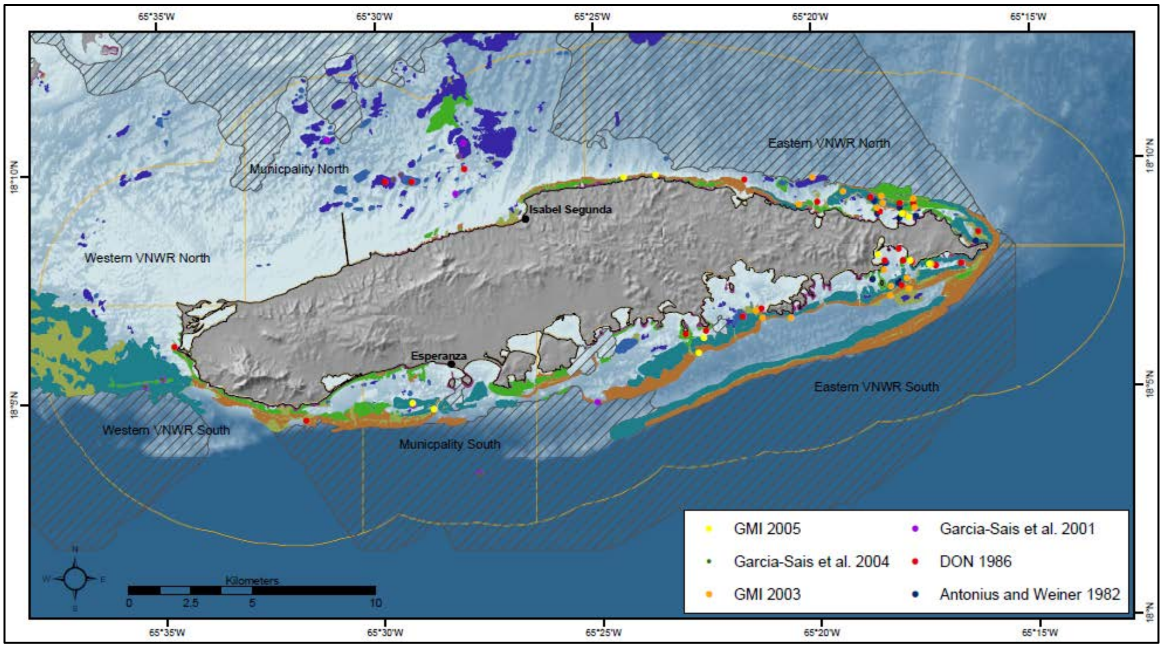


Figure 20. Reef monitoring sites where ESA-listed corals have been observed (from Bauer et al. 2008)

7.1.5 Elkhorn and Staghorn Coral Critical Habitat

There have been several efforts by NOAA's NCCOS to map and characterize the benthic habitats around Vieques (Figure 21). NCCOS's benthic habitat maps were used by the Navy as part of their characterization of specific habitat areas where anomalies that may be MEC/MPPEH were found. The Navy used NCCOS's classification scheme and determined that 5,198.2 acres within UXO 16, or 48.9 percent, contain coral reef and hard bottom. Of this acreage, hard bottom in the form of rock/boulder, pavement, and pavement with sand channels comprises 1,940.7 acres (37.3 percent) and reef in the form of aggregate reef, individual patch reef, aggregated patch reefs, and spur and groove comprises 800.8 acres (15.5 percent).

Based on underwater photographs and video collected by the Navy as part of the WAA and other survey efforts for this consultation, it is likely that the areas classified as reef within the 30 m depth limit of designated elkhorn and staghorn coral critical habitat contain the PBF. Many of the colonized bedrock, pavement, and pavement with sand channels likely also contain the PBF, though other hard bottom areas within these categories probably do not due to high sediment cover from moving sand.

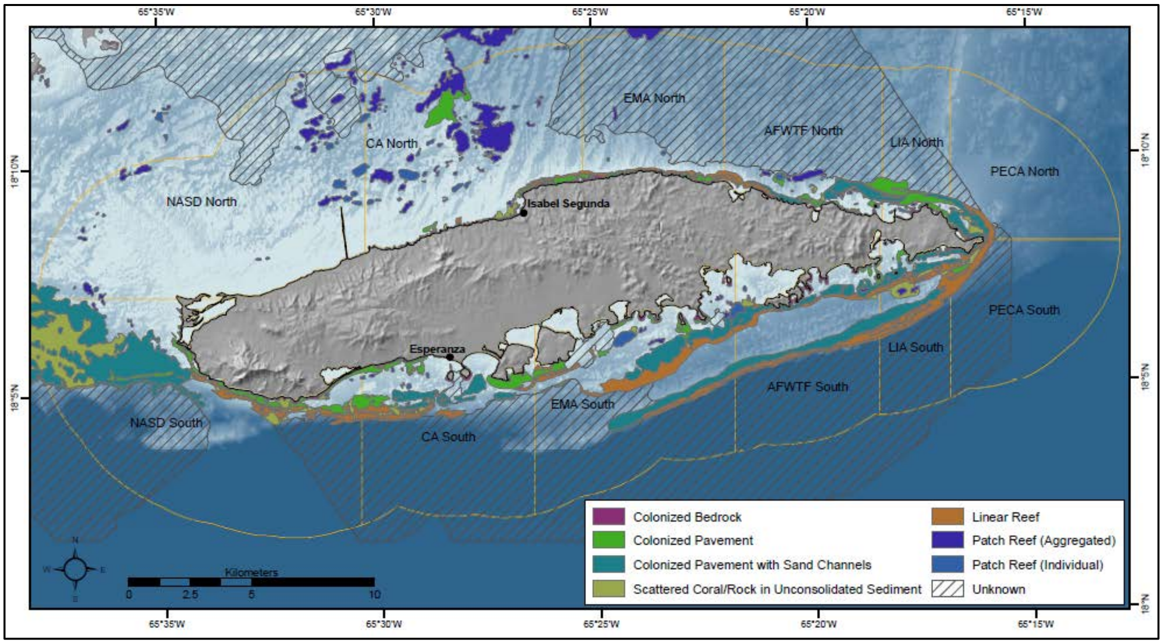


Figure 21. Distribution of hard bottom habitat around Vieques (from Bauer et al. 2008)

7.2 Factors Affecting Sperm Whales; Green, Leatherback, and Hawksbill Sea Turtles; Nassau Grouper; ESA-Listed Corals; and Elkhorn and Staghorn Coral Critical Habitat in the Action Area

7.2.1 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. Effects of climate change include sea level rise, increased frequency and magnitude of severe weather events, changes in air and water temperatures, and changes in precipitation patterns, all of which are likely to affect ESA resources. NOAA's climate information portal provides basic background information on these and other measured or anticipated climate change effects (see https://www.climate.gov).

In order to evaluate the implications of different climate outcomes and associated impacts throughout the 21st century, many factors have to be considered with greenhouse gas emissions and the potential variability in emissions serving as 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. 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 (USGCRP 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 RCP2.6, 1.1 to 2.6°C under RCP4.5, 1.4 to 3.1°C under RCP6.0, 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 (in press) 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 (Hoegh-Guldberg 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 (Hoegh-Guldberg 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).

The Atlantic Ocean appears to be warming faster than all other ocean basins except perhaps the southern oceans (Cheng et al. 2017). In the western North Atlantic Ocean, surface temperatures have been unusually warm in recent years (Blunden and Arndt 2017). A study by Polyakov et al. (2010) suggests that the North Atlantic Ocean 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 m (6,561.7 ft) 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 (Learmonth et al. 2006; Macleod 2009; Robinson et al. 2008; Kintisch and Buckheit 2006; Mcmahon and Hays 2006; Evans and Bjørge 2013; IPCC 2014). Though predicting the precise consequences of climate change on highly mobile marine species is difficult (Simmonds and Eliott 2009), 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). 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 and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

Changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, DO levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), 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). 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 and is likely to occur in the Pacific. 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).

Similarly, climate-related changes in important prey species populations are likely to affect predator populations. 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).

Macleod (2009) estimated that, based upon expected shifts in water temperature, 88 percent of cetaceans would be affected by climate change, 47 percent would be negatively affected, and 21 percent would be put at risk of extinction. Changes in core habitat area means some species are predicted to experience gains in available core habitat and some are predicted to experience losses (Hazen et al. 2012). Such range shifts could affect marine mammal and sea turtle foraging success as well as sea turtle reproductive periodicity (Silber et al. 2017; Pike 2013).

Genetic analyses and behavioral data suggest that sea turtle populations with temperature-dependent sex determination may be unable to evolve rapidly enough to counteract the negative fitness consequences of rapid global temperature change (Hays 2008 as cited in Newson et al. 2009). Altered sex ratios have been observed in sea turtle populations worldwide (Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008; Fuentes et al. 2009). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska et al. 2009). A fundamental shift in population demographics may lead to increased instability of populations that are already at risk from several other threats. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times (producing smaller hatchling), reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Fuentes et al. 2009; Fuentes et al. 2010; Fuentes et al. 2011; Azanza-Ricardo et al. 2017).

Global climate change may affect Nassau grouper. Thermal changes of just a few degrees Celsius can substantially alter fish protein metabolism (Mccarthy and Houlihan 1997), response to aquatic contaminants (Reid et al. 1997), reproductive performance (Van Der Kraak and Pankhurst 1997), species distribution limits (Mccarthy and Houlihan 1997), and community structure of fish populations (Schindler 2001). Apart from direct changes to fish survival, increased water temperatures may alter important nursery, refuge, and foraging habitats such as coral reefs. Increased ocean acidification may also have serious impacts on fish development and behavior (Raven et al. 2005), including sensory functions (Bignami et al. 2013) and fish larvae behavior that could affect fish populations (Munday et al. 2009).

In the NMFS final rule to list 20 coral species as threatened (79 FR 53851, September 10, 2014), ocean warming and acidification, associated with climate change, were identified as two of the most important threats to the current or expected future extinction risk of reef building corals.

Reef building organisms are predicted to decrease the rate at which they deposit $CaCO_3$ in response to increased ocean acidity and warmer water temperatures (Raymundo et al. 2008). Further, the most severe coral bleaching events observed to date have typically been accompanied by ocean warming events such as the El Niño-Southern Oscillation (Glynn 2001). Bleaching episodes result in substantial loss of coral cover, and result in the loss of important habitat for associated reef fishes and other biota. Corals can typically withstand mild to moderate bleaching, but severe or prolonged bleaching events can lead to coral colony death (79 FR 53851). While the susceptibility to ocean warming and acidification associated with climate change is expected to vary by species and specific coral colony (based on latitude, depth, bathymetry, etc.; 79 FR 53851), climate change is expected to have major impacts on the coral species considered in this Opinion.

Within the action area, severe hurricanes such as those during the 2017 hurricane season and severe swells such as those during the summer of 2019, coral bleaching from elevated sea surface temperatures, and sea level rise are affecting sea turtle nesting beaches and in-water habitat for the Nassau grouper, and ESA-listed corals and their designated critical habitat (Gould et al. 2018).

7.2.2 Fisheries

Commercial whalers once targeted sperm whales. Once commercial whaling ended, the species was expected to rebuild; however, a study in the eastern Caribbean indicates that unit size, numbers of calves, and calving rates in a well-studied population have continued declining (Gero and Whitehead 2016). Fishing gear used in the Caribbean, including Puerto Rico, includes gillnets, which have been shown to cause entanglement of sperm whales. Two were reported entangled in 2015 in the eastern Caribbean (Gero and Whitehead 2016). There are no reported entanglements of sperm whales in the action area, but the population in the eastern Caribbean is the same population that travels through the action area so entanglement due to fishing gear in and outside the action area could contribute to population declines.

Fishing gears used throughout the action area adversely affect threatened and endangered sea turtles. Based on stranding data from Commonwealth waters (PRDNER unpublished stranding data, net and hook-and-line gear have been documented as interacting with sea turtles in Puerto Rico. Illegal fishing targeting sea turtles accounted for 33% of reported sea turtle strandings around Vieques for the period from 1991 – 2008 with no incidental capture of sea turtles in fishing gear reported (PRDNER unpublished stranding data). All of the turtles affected by illegal fishing (i.e., harpooning) were hawksbills. Abandoned or lost fishing gear can also affect the quality of refuge and foraging habitat for green and hawksbill sea turtles as abandoned gear can lead to abrasion and breakage in hard bottom and coral reef habitats. They also have shading impacts on seagrass and macroalgae if the gear is large enough, such as traps and nets. Gear used over areas containing corals also has the potential to affect ESA-listed corals and designated critical habitat for elkhorn and staghorn corals. Stranding of sperm whales because of interactions with fisheries has not been reported in the action area and, given the artisanal nature of the fisheries in the action area in both federal and Commonwealth waters, is not likely to occur.

For all fisheries for which there is a Fishery Management Plan (FMP) or for which any federal action is taken to manage that fishery, impacts are evaluated under section 7 of the ESA. All of these opinions found that the actions described were likely to adversely affect, but not likely to jeopardize the continued existence of, sea turtle species. Formal section 7 consultations have been conducted on the Caribbean Reef Fish and Caribbean Spiny Lobster fisheries, under the jurisdiction of the Caribbean Fishery Management Council, occurring in the action area and found fisheries actions to likely to adversely affect threatened and endangered sea turtles. Anticipated levels of take associated with these actions reflect the impact on sea turtles and other listed species of each activity anticipated from the date of the ITS in the waters of the EEZ off Puerto Rico and the USVI. Anticipated levels of take under the Caribbean Reef Fish FMP are 75 lethal takes of green sea turtles over 3 years, 51 lethal takes of hawksbill sea turtles with no more than 3 non-lethal takes over 3 years, and 48 lethal takes of leatherback sea turtles over 3 years. Anticipated levels of take under the Spiny Lobster FMP are 12 lethal takes of green and hawksbill sea turtles over 3 years and 9 lethal takes of leatherback sea turtles over 3 years. Informal Section 7 consultations were also completed for the Caribbean Coral and Queen Conch FMPs. NMFS concluded that implementation of the Coral and Queen Conch FMPs are not likely to adversely affect ESA-listed sea turtles.

Anticipated levels of take are also part of section 7 consultations for FMPs in the Gulf and South Atlantic where sea turtles may be found in transit in the action area. Table 3 details the lethal and total anticipated levels of take under the Gulf of Mexico/South Atlantic Spiny Lobster and South Atlantic Snapper-Grouper FMPs and the FMPS for highly migratory species (HMS) including Coastal Migratory Pelagics, Dolphin-Wahoo, HMS-Pelagic Longline, and Shark Fisheries, as well as takes that may occur under the Southeastern U.S. Shrimp Fishery. The take numbers for the shrimp fishery were estimated based on TED enforcement as a surrogate for actual numbers of animals.

Table 3. Anticipated Levels of Take of Leatherback, Hawksbill, and Green Sea Turtles under Gulf, South Atlantic, and HMS FMPs, and the Southeastern U.S. Shrimp Fishery

FMP or Fishery	Leatherback		Hawksbill		Green	
	Lethal Takes	Total Takes	Lethal Takes	Total Takes	Lethal Takes	Total Takes
Gulf of Mexico/South Atlantic Spiny Lobster		1 (note: may be lethal or non-lethal) over 3 years		1 (note: may be lethal or non-lethal) over 3 years		3 (note: may be lethal or non-lethal) over 3 years
South Atlantic Snapper-Grouper	5	6 over 3 years	4	6 over 3 years	42 (NA DPS)	111 (NA DPS) over 3 years

					3 (SA DPS)	6 (SA DPS) over 3 years
Gulf of Mexico Reef Fish	11	11 over 3 years	8	9 over 3 years	75	116 over 3 years
Coastal Migratory Pelagics	1	1 over 3 years	1	1 over 3 years	9 (NA DPS)	31 (NA DPS) over 3 years
Dolphin-Wahoo	1	12 over 1 year	1	3 over 1 year	1	3 over 1 year
Pelagic Longline	252	1,764 over 3 years	18	105 over 3 years	18	105 over 3 years
Shark Fisheries	9	18	9	18	33	57
Southeastern U.S. Shrimp Fishery	144 per year		78 per year		1,453 per year	

Nassau grouper were an important component of the fishery and were targeted in federal and Commonwealth fisheries until fishing was prohibited (in federal waters in 1990 and in Commonwealth waters in 2004). Fishing in Commonwealth waters occasionally targeted juveniles in nearshore areas in addition to adults. As the fishery became more diminished, younger life stages were targeted, leading to the prohibition of fishing for this species year-round in federal and Commonwealth waters.

Several types of fishing gear may also adversely affect coral colonies and critical habitat. Longline, other types of hook-and-line gear and traps have all been documented as interacting with coral habitat and coral colonies in general, though no data specific to ESA-listed corals and their habitat is available. Available information suggests hooks and lines can become entangled in reefs, resulting in breakage and abrasion of corals. Net fishing can also affect coral habitat and coral colonies if this gear drags across the marine bottom either due to efforts targeting reef and hard bottom areas or due to derelict gear. Studies by Sheridan et al. (2003) and Schärer et al. (2004) showed that most trap fishers do not target high-relief bottoms to set their traps due to potential damage to traps. Unfortunately, lost traps and illegal traps can affect corals and their habitat if they are moved onto reefs or colonized hard bottoms during storms or placed on coral habitat because the movement of the traps leads to breakage and abrasion of corals.

NMFS reinitiated section 7 consultations for the Coral, Queen Conch, Reef Fish, and Spiny Lobster FMPs under the jurisdiction of the CFMC when elkhorn and staghorn corals were listed and critical habitat was designated for these corals. NMFS concluded that the implementation of the Coral FMP would have no effect on listed corals or coral designated critical habitat. NMFS then reinitiated consultation again for the Spiny Lobster and Reef Fish FMPs on September 26, 2016 because of the 2014 listing of pillar, rough cactus, lobed star, mountainous star, and boulder star corals. On January 19, 2016, NMFS determined the authorization of fishing managed by the Spiny Lobster and Reef Fish FMPs was not likely to adversely affect these corals.

Commonwealth-managed fisheries operating in the action area have potential impacts to sea turtles, ESA-listed corals, and coral habitat similar to those analyzed in the CFMC FMP consultations described above. Commonwealth waters extend to 9 nautical miles from the shore meaning they encompass shallow and deep water areas where all three sea turtle species, all seven ESA-listed coral species, and elkhorn and staghorn coral critical habitat (to 30 m depth) may be present. As noted above, there was no incidental catch of sea turtles in fishing gear in the action area for the period from 1991 – 2008, but sea turtle poaching (i.e, targeting hawksbills) is common, accounting for 33% of strandings (PRDNER unpublished stranding data). There are active commercial fishing communities in Isabela Segunda and Esperanza communities in Vieques, as well as on the east coast of Puerto Rico, and there are recreational fishers.

7.2.3 Vessel Operation and Traffic

Potential sources of adverse effects from federal vessel operations in the action area include operations of NOAA vessels, anchor and propeller damage and accidental groundings. NOAA, including NOS and other line offices, conduct coral reef monitoring, benthic surveys, sediment sampling and other scientific surveys in the action area. NOS and the Southeast Fishery Science Center lead the NOAA NCRMP efforts that take place every 2 years at randomly selected sampling sites around Puerto Rico. NOAA's Coral Reef Conservation Program (CRCP) has requested initiation of a programmatic ESA section 7 consultation for the monitoring program and other efforts that receive some or all of their funding from the coral program with NMFS's Office of Protected Resources. EPA conducts coral surveys at different locations around Puerto Rico, often annually. In the past, EPA used a large research vessel to complete these surveys. However, the agency no longer owns the vessel so coral survey operations are done using smaller motorized vessels, typically through rental agreements with local operators. EPA has not initiated an ESA section 7 consultation for their coral survey program at this time.

NMFS and the U.S. Coast Guard (USCG) completed an informal programmatic section 7 consultation for the Caribbean Marine Event Program for marine events in USVI and Puerto Rico in December 2017. As a result of this consultation, the USCG includes guidelines to avoid and minimize potential impacts of marine events, especially events involving motorized vessels such as speedboat races, to ESA-listed species and their habitat as permit conditions the event participants must follow. NMFS has also completed a formal consultation with the USCG to cover maintenance of federal ATONs throughout Puerto Rico and the rest of District 7. ATON maintenance requires the use of USCG cutters and the consultation included requirements to minimize potential impacts of vessel operation and other actions associated with ATON maintenance on ESA-listed corals and their habitat. ATONs are present in some portions of the action area, particularly ports and dock areas, including those used by the Navy on Vieques such as in Esperanza.

Through the Section 7 process, where applicable, NMFS will establish conservation measures for federal agency vessel operations to avoid or minimize adverse effects to ESA-listed species in the action area from vessel transit, anchoring, and other vessel operations. However, vessel operation do present the potential for some level of interaction with ESA-listed species in the action area.

Commercial and recreational vessel traffic can have adverse effects on sperm whales, ESA-listed sea turtles and corals and their habitat via propeller injuries and boat strike injuries (turtles), and accidental groundings, propeller scarring, and propeller wash (corals and habitat for sea turtles and corals). NMFS did not find records of vessel collisions with sperm whales but, because deeper waters of the action area include routes for shipping traffic, there is a possibility of vessel collision, some of which may be unreported. PRDNER stranding data indicate that 13 green sea turtles and 16 hawksbill sea turtles could be confirmed to have been impacted by boats in the action area from 1989-2009. The proliferation of vessels is associated with the proliferation and expansion of docks, the expansion and creation of port facilities, and the expansion and creation of marinas. While the construction of facilities is limited in most of UXO 16 in areas offshore of VNTR lands, the Navy has documented the proliferation of vessels in some nearshore areas in UXO 16. In Bahía Icacos on the northeast coast of the VNTR, the Navy recorded 81 vessels in the bay in 2005, 59 in 2006 and 2007, 991 in 2008, 627 in 2010, and 622 from January to October 2011 (CH2M Hill 2011). The action area also includes the east coast of Puerto Rico where port and marina expansion and dock construction occur and other areas around Vieques that are not federally managed. As part of the section 7 consultation for dock, port, and marina construction activities under the jurisdiction of the U.S. Army Corps of Engineers (USACE), NMFS also considers the impacts of vessel traffic from the operation of these facilities and any measures to avoid and minimize adverse impacts to sea turtles. Additionally, because the construction of many of these in-water facilities involves pile driving, NMFS also considers the potential acoustic impacts of facility construction on marine mammals, sea turtles, and fish and any measures to avoid and minimize injurious and behavioral acoustic impacts to these animals.

Commercial and recreational vessel traffic in the action area is also associated with commercial and private diving activities. There are several areas around Vieques that are visited by commercial dive operations from Vieques and the east coast of Puerto Rico and by private individuals. Anchoring of these vessels at reef sites can lead to impacts to corals and habitat used by ESA-listed sea turtles and corals.

7.2.4 Research Activities

Regulations developed under ESA section 10(a)(1)(A) allow for the issuance of permits authorizing take of certain ESA-listed species for the purpose of scientific research. In addition, section 6 of the ESA allows NMFS to enter into cooperative agreements with states and territories to assist in recovery actions for listed species. Prior to issuance of any section 10 permit, the proposal must be reviewed for compliance with section 7 since NMFS is the action agency. Sperm whales, sea turtles, and elkhorn and staghorn corals have “take” prohibitions due to their listing as endangered or the promulgation of a 4(d) rule. For elkhorn and staghorn coral, the 4(d) rule enables permits issued by the Commonwealth to be used in lieu of section 10 permits issued by NMFS for activities meant to promote scientific research, enhancement, and recovery of these two coral species. PRDNER has coral monitoring sites around Vieques that have been funded by a section 6 grant, as well as by NOAA's CRCP. PRDNER has also held permits from NMFS for conducting research on various life stages of green and hawksbill sea turtles at locations around Puerto Rico, including Vieques. NMFS Southeast Fishery Science Center (SEFSC) has also held permits from NMFS OPR for conducting research on all ESA-listed sea turtle species in the Atlantic Ocean, Gulf of Mexico, and the Caribbean Sea, though the majority of their research is not conducted in the U.S. Caribbean.

In addition to authorization under the ESA, the MMPA requires that researchers obtain authorization for directed and incidental take of marine mammals. The issuance of these authorizations (under both the ESA and MMPA), often require section 7 consultation with NMFS OPR by the NMFS Permits and Conservation Division so many of the permits identified above have also undergone section 7 consultation.

MMPA authorizations include one for the Navy (that expired in July 2019) to conduct research on marine mammals, including sperm whales, in the Atlantic Ocean, Caribbean Sea, Gulf of Mexico, and Sargasso Sea, and an incidental take authorization for the SEFSC to take marine mammals incidental to fisheries research in the Atlantic Ocean, Gulf of Mexico, and Caribbean Sea.

CRCP has also funded survey work by NCCOS to evaluate benthic habitats and fish in areas around Vieques. In addition, the NCRMP randomly selects sites to survey every other year in Puerto Rico and sites can include areas around Vieques or off the east coast of Puerto Rico that are within the action area. However, survey work by NCCOS and under the NCRMP is non-intrusive so impacts to ESA-listed species, particularly sea turtles and corals, if they occur at all, would be minor and short-term from diver operations.

7.2.5 Coastal and Marine Development

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local, or private action, may indirectly affect sperm whales, sea turtles, ESA-listed corals, and elkhorn and staghorn coral critical habitat in the action area. However, because sperm whales are not permanent residents in the action area and are an offshore species, these affects are not likely to be measurable.

Sources of pollutants in the action area include atmospheric loading of pollutants, stormwater runoff from coastal towns, and runoff into water bodies that empty into bays and groundwater. However, because the project is located within the former VNTR, development has not occurred in this area. Development is not expected to occur in the future in much of UXO 16 where waters are adjacent to lands managed within the USFWS National Wildlife Refuge. Activities that may result or may have already resulted in marine pollution in the action area are related to military practices and cleanup efforts to remove surface and subsurface MEC from land and water, which are discussed in Section 8.2.6.

Coastal runoff, marina and dock construction, dredging, increased underwater noise, and boat traffic can degrade marine habitats used by sea turtles, where ESA-listed corals and coral habitat may also be present and affected by in-water activities. Many of these activities will be limited in areas of UXO 16, because they now comprise parts of the USFWS National Wildlife Refuge, but other locations in the action area, such as along the east coast of Puerto Rico and in the two towns in Vieques, have been experiencing increases in in-water construction and boating. In addition, the departure of the Navy from Vieques has resulted in an increase in tourism development on the island, including hotels, houses, and marine facilities. An increase in the number of docks built increases boat and vessel traffic. Fueling facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive coastal habitats. Although these contaminant concentrations do not likely affect the more pelagic waters, the species of sea turtles analyzed in this Opinion travel between nearshore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles.

There are studies on organic contaminants and trace metal accumulation in green and leatherback sea turtles (Aguirre et al. 1994; Caurant et al. 1999). 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 turtle size were observed in green turtles, most likely attributable to a change in diet with age. Sakai et al. (1995) found the presence of metal residues occurring in loggerhead turtle organs and eggs. Storelli et al. (1998) analyzed tissues from twelve 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. Similarly, limited data are available for ESA-listed corals related to exposure and toxicity thresholds for things like heavy metals. Exposure data that are available, such as from studies using mountainous star coral indicate that chronic exposure to certain concentrations of copper result in effects to embryo development (Bielmyer et al. 2010). Contaminants such as heavy metals used in Vieques as part of the management of the VNTR and discarded within the VNTR and the NASD could reach nearshore waters if coastal development were to occur, particularly on lands outside the USFWS refuge that were part of the NASD, affecting ESA-listed sea turtles and corals.

7.2.6 Military Activities

The eastern portion of Vieques Island was part of the VNTR. The VNTR provided logistics support, scheduling assistance, and facilities for naval gunfire support and air-to-ground ordnance delivery training for Atlantic Fleet ships (CH2M Hill 2011). Ship, air, and ground forces carried out training activities within the VNTR, including artillery and tank fire, ordnance delivery, air-to-surface mine delivery, amphibious landings, and small arms fire. Several point and area targets for ships were located within the VNTR and munitions and explosives of concern (MEC) until live fire training stopped in 2003 affected the entire 900 acres of the LIA. There is a report from May 1995 of a leatherback sea turtle with apparent acoustic trauma stranding in Vieques that the PRDNER speculated could have occurred during military training activities (PRDNER unpublished stranding data), but there is no evidence linking the turtle mortality with actual training activities. Some coral areas were used as targets, in addition to being affected by skips and misses during live fire exercises, indicating that past training activities resulted in impacts to ESA-listed corals and their habitat. Records of in-water ordnance hits and ordnance locations resulting from skips and misses in 2001 within the LIA superimposed on a map of reef habitat show two areas of probable impacts on coral reefs and seagrass: the eastern side of Bahía Salinas del Sur (in-water ordnance hits) and a relatively large area off the north side of the LIA (concentrated ordnance locations) between Punta Gato and Punta Salinas (GMI 2003). It is possible that the use of in-water targets could have affected sperm whales, likely by causing them to move away from areas within the radius of behavioral acoustic effects during in-water use of explosives but there were no surveys looking for whales conducted during these training activities that would enable a determination of whether or not effects to sperm whales occurred.

The Navy initiated a time critical removal action in 2005 in the LIA to address surface MEC on land. Cleanup activities are ongoing on land and involve vegetation clearing, digging, and detonations of accumulated items. Terrestrial work has been done on slopes and beaches and there have been some reports of stormwater transport of sediment during rain events due to the steep slopes that are now denuded of vegetation in some areas where cleanup has occurred.

As part of future cleanup activities or management of the refuge, it may be necessary to construct temporary or permanent docks or other structures in the water. Vessel use in the area is a problem that the Navy is trying to address through the installation of the barriers. These activities in nearshore waters can negatively affect ESA-listed species and their habitats.

The Navy has completed surveys to identify the presence of underwater MEC/MPPEH in UXO 16, including through the WAA. In SWMU 4, the Navy conducted an underwater site inspection in July 2012, and expanded site inspection in April-May 2015, and sediment sampling to determine whether contamination of nearshore sediment has occurred because of past disposal activities in the lagoon within SWMU 4 in 2016. Based on reports and participation in surveys by NMFS biologists, disturbance of benthic habitats in the nearshore area of SWMU 4 related to this work has been minimal.

The Navy has conducted NTCRA from in-water areas adjacent to Cayo la Chiva (June 2017) and UXO 15 PL-9 East adjacent to UXO 16 (encrusted munitions, March 2018). All of these removals were conducted following the SOPs developed by the Navy in coordination with NMFS, none involved damage to live ESA-listed coral colonies or their habitat, and seagrass disturbance was minimal.

The Navy (Space and naval Warfare Systems Center Pacific) with support from the USACE and Oklahoma State University completed a technology demonstration in Bahía Salinas del Sur of a continuous sampling approach to sampling for polar organic compounds called Polar Organic Chemical Integrative Samplers (POCIS). The bay was gridded and the magnitude and frequency of detected MC using time-weighted average concentrations derived from POCIS were compared with grab samples and aquatic toxicity screening values for MC. A second set of POCIS canisters were deployed adjacent to munitions suspected of potentially leaking MC. Focused sediment sampling was also conducted at four stations where RDX (Royal Demolition eXplosive or cyclonite or hexogen) detection was above method reporting limits to assess the relative usefulness of POCIS as a screening tool for water and sediment MC contamination (Rosen et al. 2017). This type of sampler deployment is part of the sampling activities that would be part of the proposed action. The sampler has a very small bottom footprint in terms of anchoring it to the substrate and installation and removal did not result in measurable impacts to benthic habitats.

In addition, in order to minimize the potential for interactions between boaters and MEC/MPPEH in in-water areas around the VNTR, the Navy installed mariner warning buoys off the LIA in 2007 (that have since been lost, ESA section 7 consultation concluded May 2, 2007, reference SER-2007-1856) and around Cayo la Chiva in 2012 (ESA section 7 consultation concluded July 23, 2012, reference SER-2012-2335; buoys removed in 2017 because NTCRA was concluded). The Navy also installed a series of barriers to prevent large commercial and recreational vessels from accessing channels in Bahía Icacos in 2012 (ESA section 7 consultation concluded August 20, 2012, reference SER-2011-5676) one of which failed due to oceanographic conditions and the rest of which were removed in 2017. The formal consultation for the barrier system required that areas containing ESA-listed corals and their habitat be avoided when installing the barrier anchors, that the footprint of anchors in seagrass beds be minimized, and that reef marker buoys be installed using anchor systems with minimal footprints in shallow reef and seagrass areas to minimize the potential for accidental groundings of vessels trying to go around the barriers. The Navy is looking for alternatives to the barriers or to reinstall a similar barrier system in Bahía Icacos, which would be part of the in-water structure installations covered under this Opinion.

7.2.7 Natural Disturbance

Hurricanes and large coastal storms can significantly alter habitats used by ESA-listed sea turtles and corals. These storms can also directly affect sperm whales, sea turtles, and ESA-listed corals. The movement of whales and sea turtles can be affected by oceanographic conditions caused by large storms and, for species such as sperm whales and leatherback sea turtles that forage offshore, can shift locations of prey species. In addition, early life stages of sea turtle species can be transported by currents and waves to areas that are not suitable for the animals or where they cannot find adequate food, leading to mortality. Waves and currents can also cause breakage and overturn coral colonies, as well as deposit sediment and debris on colonies, leading to breakage and abrasion.

Historically, large storms potentially resulted in asexual reproductive events, particularly for branching coral species, if the fragments encountered suitable substrate, attached, and grew into new colonies. However, recently, the amount of suitable substrate has been significantly reduced; therefore, many fragments created by storms die. Hurricanes are also sometimes beneficial, if they do not result in heavy storm surge, during years with high sea surface temperatures, as they lower temperatures providing fast relief to corals during periods of high thermal stress (Heron et al. 2008). This reduction in temperature also benefits hawksbill sea turtles because the sponge species they prefer to eat can suffer from thermal stress and bleach or die.

Major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs in Puerto Rico, as well as loss or damage to seagrass beds from blowouts and sediment movement. Tropical storms and hurricanes can result in severe flooding, leading to significant sediment transport to nearshore waters from terrestrial areas, as well as shifting of marine sediments. In addition to affecting sessile benthic organisms such as ESA-listed corals, changes in the structure of the reef affect species like sea turtles, in particular greens and hawksbills that use reef habitats for refuge and foraging. In-water habitat for green and hawksbill sea turtles is temporarily or permanently lost or degraded depending on the magnitude of the storm.

Based on NOAA hurricane data and data from the Federal Emergency Management Agency, there have been a total of 11 hurricanes and tropical storms that have affected Puerto Rico between 1975 and 2017. Hurricane David in 1979 caused extremely violent sea conditions along the south coast of the island and severe flooding across the island and on associated islands including Vieques. Hurricane David was followed five days later by Tropical Storm Frederick resulting in additional flooding. Hurricane Hugo in 1989 also led to violent sea conditions and major flooding across the island and associated islands. Hurricanes Marilyn (in 1995) and Hortense (in 1996), though not as intense, led to additional impacts to reefs and seagrass beds already suffering damage from Hurricane Hugo. When Hurricane Georges hit Puerto Rico in 1998, many nearshore marine habitats had already been impacted by previous storms and associated land-based sources of pollution due to flooding. Hurricane Irene in 2011 affected the north and northeast coasts of Puerto and associated islands including Vieques through extremely violent sea conditions and flooding.

Hurricanes Irma and Maria passed through the Caribbean in September 2017. Many portions of Puerto Rico were relatively unaffected by Hurricane Irma, although the storm did cause damage to Vieques, but Hurricane Maria affected all of Puerto Rico. The islands are still recovering from the effects of the storms but in-water assessments of habitats indicate that some coral areas suffered only minor damage from the storm while other areas suffered significant damage (Figure 22). In other areas, triage of affected corals was performed to stabilize colonies affected by the storms and work on reef restoration is still on going. Seagrass beds also suffered varying levels of effects depending on their location around the islands in relation to currents, waves, and storm surge. Reports of impacts from Hurricanes Irma and Maria on coastal areas of Puerto Rico indicate that beaches in many parts of the island and outlying islands such as Culebra and Vieques were significantly affected by erosion associated with storm surge (E. Díaz, PRDNER, pers. comm. to L. Carrubba, NMFS, October 12, 2017). There were also reports of numerous vessel groundings, contamination of nearshore waters due to flooding of terrestrial areas including wastewater treatment plants, transport of debris to nearshore waters and debris accumulations where in-water structures were damaged, and storm damage to coral and seagrass habitats (E. Díaz, PRDNER, pers. comm. to L. Carrubba, NMFS, October 12, 2017). Some benthic habitats that did not suffer physical impacts from the hurricanes are not fully recovering apparently due to the longer-term effects of contaminant and debris transport to nearshore waters associated with flooding caused by the storm.

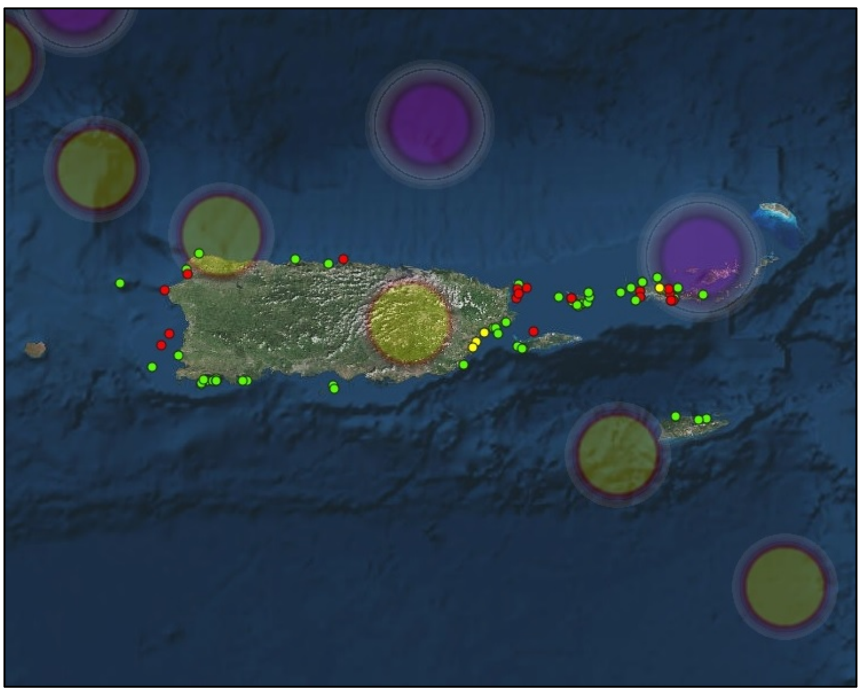


Figure 22. Map showing tracks of Hurricanes Irma (large purple dots) and Maria (large yellow dots) in area where Puerto Rico is located and results of coral surveys conducted through 2018. Small green dots indicate areas where coral surveys indicated no triage was needed, red dots indicate areas where triage was needed, and yellow dots indicate areas where the need for triage was still under evaluation (NOAA Restoration Center,

https://noaa.maps.arcgis.com/apps/MapJournal/index.html?appid=4f7e03fe4c3748849426d15e12491d22)

7.3 Synthesis of Baseline Impacts

Collectively, the stressors described above have had, and are likely to continue to have, lasting impacts on sperm whales; green (North and South Atlantic DPS), leatherback, and hawksbill sea turtles; Nassau grouper; ESA-listed corals; and elkhorn and staghorn coral critical habitat within the action area. Some of these stressors, such as fishing, result in mortality or serious injury to individual animals, whereas others result in more indirect (e.g., water quality degradation from coastal development) or non-lethal (e.g., research permits involving only observation of marine mammals) impacts.

We consider the best indicator of the environmental baseline on ESA-listed resources to be the status and trends of those species. As noted in Section 6.2, some of the species considered in this consultation appear to have stable populations, others are declining, and for others, their population trends remain unknown. Taken together, this indicates the environmental baseline is affecting species in different ways. The species with stable populations are not declining despite the potential negative impacts of the environmental baseline. Therefore, while the baseline may slow their recovery, recovery is not being prevented. For the species that may be declining in abundance, it is possible that the suite of conditions described in this *Environmental Baseline* section is limiting their recovery. However, it is also possible that their populations are at such low levels (such as for Nassau grouper, which was at the level of commercial extinction by 1986 in the U.S. Caribbean) that even when the species' primary threats are removed, the species may not be able to achieve recovery. At small population sizes, species may experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their limited population size to become a threat in and of itself. A thorough review of the status and trends of each species for which NMFS has found the action is likely to cause adverse effects is discussed in *Status of Species and Critical Habitat Likely to be Adversely Affected* (Section 6.2) of this Opinion.

8 EFFECTS OF THE ACTION

"Effects of the action" has been recently revised to mean all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur (50 C.F.R. §402.02). Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (see 50 C.F.R. §402.17).

This effects analyses section is organized following the stressor, exposure, response, risk assessment framework.

8.1 Discountable and Insignificant Effects

We have determined that sperm whales; green, leatherback, and hawksbill sea turtles, Nassau grouper; ESA-listed corals, and elkhorn and staghorn coral critical habitat may be adversely affected by the proposed action. However, some of the effects of stressors from the proposed action (Section 5) to these species and designated critical habitat will be discountable or insignificant and therefore not likely to result in adverse effects. These stressors are discussed below.

8.1.1 Strikes/Collisions

Vessel operations associated with all of the activities that are part of the proposed action could lead to collisions with sperm whales, and green, hawksbill, and leatherback sea turtles. However, an analysis of 354 known sea turtle strandings for all of Puerto Rico from 1989-2009 revealed that the sea turtles that had injuries consistent with vessel collisions did not include any strandings around Vieques (PRDNER unpublished stranding data). There are no reports of vessel collisions with sperm whales. In addition, the Navy has been conducting in-water survey and cleanup activities in UXO 16 since approximately 2004 and has not reported sightings of sperm whales or any vessel collisions with whales or sea turtles. Therefore, we believe that the effects of vessel collisions associated with the proposed action on sperm whales and ESA-listed sea turtles will be discountable and thus not likely to adversely affect these species. Vessel collisions are expected to have no effect on Nassau grouper because these fish do not need to surface to breathe and larger individuals that could be struck by vessels prefer to be in deeper water.

Activities associated with the location and removal of MEC/MPPEH and underwater investigations also have the potential to result in collisions with vessels and/or equipment for sperm whales, green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals. MEC/MPPEH items that are thought to be unstable and must be removed remotely require towing of the items. The majority of removal operations are expected to occur in nearshore waters along coastlines and within embayments where sperm whales would not be present. Sperm whales and sea turtles are expected to be observed by vessel crew members and/or divers engaged in removal activities and sea turtles are expected to move away from these activities in response to the noise and movement associated with them. Large Nassau grouper are also expected to move away from the disturbance associated with removal activities, but given the rarity of large groupers, it is unlikely that any will be present during removal activities. Juveniles are more likely to be in nearshore waters and would swim away from any disturbance associated with removal activities. Similarly, collisions with ROVs and remote sensing and other towed equipment are also possible. The Navy has been performing survey work using ROVs and towed equipment for a number of years within UXO 16, including in order to complete the WAA, and the only reported collision was with the marine bottom in an area containing ESA-listed coral colonies. Therefore, we believe that the effects of collisions associated with MEC/MPPEH removal activities and the use of ROVs and towed equipment to sperm whales, green and hawksbill sea turtles, and Nassau grouper will be discountable and thus not likely to adversely affect these species.

The effects of collisions with ESA-listed corals are discussed in Section 8.2.

8.1.2 Vessel Anchoring, Propeller Wash and Scarring, and Groundings

Vessel anchoring and propeller wash and scarring could affect in-water habitats used by green and hawksbill sea turtles and Nassau grouper, including seagrass beds and coral habitats, which are abundant in UXO 16. Vessel anchoring and impacts from propellers being operated in water depths that are not appropriate for the vessel draft or in areas with coral heads close to the water surface could affect ESA-listed corals and elkhorn and staghorn coral critical habitat. However, many of the vessels used by the Navy and its contractors for work in UXO 16 have shallow drafts and the Navy has developed a number of SOPs in coordination with NMFS for in-water surveys that include vessel operation to minimize potential impacts to benthic habitats. A NMFS biologist was present during field operations in nearshore waters of SWMU 4 in July 2012, observed that the anchor of the contractor's vessel was in an area containing coral, and was caught on a non-ESA-listed coral. A diver freed the anchor and the vessel was relocated to an area with sand bottom. An SOP was developed, which is included in the PDCs, to insure divers check areas where vessels will anchor to verify that no coral habitats are present. We believe the effects of vessel anchoring and propeller wash and scarring on green and hawksbill sea turtles and Nassau grouper due to habitat impacts will be insignificant due to the SOPs employed by the Navy to minimize potential impacts to habitats from vessel operations and the extent of seagrass and coral habitats within UXO 16. We believe the effects of vessel anchoring and propeller wash and scarring on ESA-listed corals and elkhorn and staghorn coral critical habitat will be discountable due to the SOPs to minimize potential impacts to coral species and their habitats. Therefore, vessel anchoring and propeller wash and scarring are not likely to adversely affect green and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat. Sperm whales do not use shallow water habitat; therefore, vessel anchoring and propeller wash and scarring will have no effect on these animals. Similarly, leatherbacks do not use nearshore benthic habitat other than as transit areas during nesting season as this species is pelagic and forages on prey in ocean waters; therefore vessel anchoring and propeller wash and scarring will have no effect on these animals.

A review of approximately 170 vessel grounding reports from the U.S. Caribbean prepared by the USCG from 2016 through September 2019 revealed three reported vessel groundings around Vieques, one of which was a vessel that was lost during Hurricane Maria and ended up on a reef in Vieques. While all of these groundings apparently affected coral reefs likely to contain ESA-listed corals and elkhorn and staghorn coral critical habitat, none of the vessels were associated with activities carried out by the Navy and its contractors in UXO 16. Therefore, we believe the effects of vessel grounding associated with the proposed action on ESA-listed corals and elkhorn and staghorn coral will be discountable and thus not likely to adversely affect listed coral species or critical habitat.

8.1.3 **Accidental Spills and Marine Debris**

NOAA's ResponseLink (https://responselink.orr.noaa.gov) documented three incidents involving vessels, two of which had the potential to cause an oil spill. One was associated with a vessel collision between a pleasure craft and a fishing vessel in September 2014 with both vessels having 200 to 250 gallons of fuel on board and is not reported to have resulted in an oil spill. The other was associated with the grounding of Merchant Vessel (M/V) Ferrel, which was lost during Hurricane Maria and was first reported as a hazard in October 2017. M/V Ferrel ended up stranded within UXO 16 and a lightering operation was performed in February 2018 to remove approximately 7,800 gallons of fuel from the vessel's tanks in order to minimize the potential for an oil spill. No oil spills have been reported because of ongoing Navy survey and cleanup operations in UXO 16.

In addition to accidental spills, vessel regularly discharge into marine waters as part of normal operations. Discharges include deck runoff, leaching of antifouling products, bilgewater, and other waste streams, which vary depending on the size and type of vessel. Some of the vessels used by the Navy and its contractors as part of the proposed action may have toilets, kitchens, showers, or other sources of discharges. However, the majority of vessels used to conduct the activities that are part of the proposed action are small vessels such as zodiacs with only a center console. Vessel motors often discharge a small amount of petroleum products during normal operation as well. There are regulations (largely under the authority of EPA) governing the location of certain discharges, such as sanitary wastewater, and require controls for some discharges that contain contaminants to minimize their release into marine waters. In addition, most of the vessels used during the proposed activities are removed from the water each day once work is completed, reducing the potential for long-term exposure of ESA-listed species and their habitats to discharges such as antifouling leachate. Vessels also generate marine debris such as lost equipment and trash that falls into the water. Because divers are used in the majority of activities that are part of the proposed action, any equipment or gear that falls in the water during operations can be retrieved. Gear and equipment is stored while underway, which also reduces the potential for items to fall into the water. Because most work does not involve overnight stays on the water and some work is done from the shoreline, trash generation is minimal.

Based on the above, we believe the effects of accidental spills, vessel discharges, and marine debris on sperm whales, green, leatherback and hawksbill sea turtles, Nassau grouper, ESA-listed coral, and elkhorn and staghorn coral critical habitat will be discountable and thus not likely to adversely affect these species and critical habitat.

8.1.4 **Vessel and Equipment Noise**

The effects of noise from nonintentional detonation and BIPS are discussed in Section 8.2.

Vessel Noise

Underwater sound from vessels is generally at relatively low frequencies, usually between 5 and 500 Hz (Hildebrand 2009; NRC 2003; Urick 1983; Wenz 1962; Southall et al. 2017). Low frequency ship noise sources include propeller noise (cavitation, cavitation modulation at blade passage frequency and harmonics, unsteady propeller blade passage forces), propulsion machinery such as diesel engines, gears, and major auxiliaries such as diesel generators (Ross 1976). High levels of vessel traffic are known to elevate background levels of noise in the marine environment (Andrew et al. 2011; Chapman and Price 2011; Frisk 2012; Miksis-Olds et al. 2013; Redfern et al. 2017; Southall 2005). Anthropogenic sources of vessel noise include recreational vessel, small commercial fishing vessels, vessels for tourism and scientific research, and some larger vessels such as cargo ships that may transit between Vieques and the main island of Puerto Rico. These vessels produce varying noise levels and frequency ranges. Commercial ships radiate noise underwater with peak spectral power at 20–200 Hz (Ross 1976). The dominant noise source is usually propeller cavitation which has peak power near 50–150 Hz (at blade rates and their harmonics), but also radiates broadband power at higher frequencies, at least up to 100,000 Hz (Arveson and Vendittis 2000; Gray and Greeley 1980; Ross 1976). While propeller singing is caused by blades resonating at vortex shedding frequencies and emits strong tones between 100 and 1,000 Hz, propulsion noise is caused by shafts, gears, engines, and other machinery and has peak power below 50 Hz (Richardson et al. 1995). Overall, larger vessels generate more noise at low frequencies (<1,000 Hz) because of their relatively high power, deep draft, and slower-turning engines (<250 rotations per minute) and propellers (Richardson et al. 1995). However, none of these vessels would be associated with the proposed action.

One potential effect from vessel noise is auditory masking that can lead animals to miss biologically relevant sounds that species may rely on, as well as eliciting behavioral responses such as an alert, avoidance, or other behavioral reaction (NRC 2003;2005; Williams et al. 2015). There can also be physiological stress from changes to ambient and background noise. The effects of masking can vary depending on the ambient noise level within the environment, the received level, frequency of the vessel noise, and the received level and frequency of the sound of biological interest (Clark et al. 2009; Foote et al. 2004; Parks et al. 2010; Southall et al. 2000). In the open ocean, ambient noise levels are between about 60 and 80 dB re: 1 μPa, especially at lower frequencies (below 100 Hz; NRC 2003). When the noise level is above the sound of interest, and in a similar frequency band, auditory masking could occur (Clark et al. 2009). Any sound that is above ambient noise levels and within an animal's hearing range needs to be considered in the analysis. The degree of masking increases with increasing noise levels. A noise that is just detectable over ambient levels is unlikely to cause any substantial masking above that which is already caused by ambient noise levels (NRC 2003;2005).

Given that the range of best hearing for ESA-listed sea turtles appears to be 100 to 400 Hz and between 300 to 1000 Hz for fishes (including elasmobranchs, although limited information is available for groupers), the frequency range for operation of small vessels is outside the hearing range of sea turtles and Nassau grouper so noise from operation of small vessels is not expected to affect these animals. Closer interactions with vessels and ESA-listed sea turtles may elicit avoidance behavior such as diving and fast swimming, which may result in short interruptions in feeding and other behaviors (NMFS 2018a).

The hearing range of marine mammals is highly variable. Unlike ESA-listed sea turtles and fish, sperm whales are likely to detect a range of sounds, including motor noise from small vessels. Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. Most of the investigations reported that animals tended to reduce their visibility at the water's surface and move horizontally away from the source of disturbance or adopt erratic swimming strategies (Corkeron 1995; Lundquist et al. 2012; Lusseau 2003;2004; Nowacek et al. 2001; Van Parijs and Corkeron 2001; Williams et al. 2002b; Williams et al. 2002a). In the process, their dive times increased, vocalizations and surface-active behaviors were reduced (with the exception of beaked whales), individuals in groups moved closer together, swimming speeds increased, and their direction of travel took them away from the source of disturbance (Baker and Herman 1989; Edds and Macfarlane 1987; Evans et al. 1992; Kruse 1991). Some individuals also dove and remained motionless, waiting until the vessel moved past their location. Several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). Although many studies focus on small cetaceans, studies of large whales have reported similar results for fin and sperm whales (David 2002). Sperm whales generally react only to vessels approaching within several hundred meters; however, some individuals may display avoidance behavior, such as quick diving (Magalhaes et al. 2002; Wursig et al. 1998). One study showed that after diving, sperm whales showed a reduced timeframe from when they emitted the first click than before vessel interaction (Richter et al. 2006).

Based on available information and other consultations such as those for the use of military vessels in training and testing activities, we conclude that sperm whales, green and hawksbill sea turtles, and Nassau grouper in the action area are likely to either not react or to exhibit avoidance behavior in response to vessel noise and movement. Most avoidance responses would consist of movements away from vessels, perhaps accompanied by slightly longer dives by sperm whales and turtles (NMFS 2015a). Most of the temporary changes in behavior would consist of a shift from behavioral states with low energy requirements like resting, to states with higher energy requirements like active swimming, with the animals then returning to the lower energy behavior. For behavioral responses to result in energetic costs that result in long-term harm, such disturbances would likely need to be sustained for a significant duration or extent, which is not expected for activities that are part of this consultation. Thus, we do not expect sperm whales, green and hawksbill sea turtles, and Nassau grouper to respond to vessel noise or to respond measurably to vessel transit in ways that would significantly disrupt normal behavior patterns including breeding, feeding, or sheltering. Therefore, we believe the effects of noise from vessel operation associated with the proposed action will be insignificant and thus not likely to adversely affect these animals.

Equipment Noise

Echosounders may be used by vessels to aid in navigation. An echosounder measures the round trip time it takes for a pulse of sound to travel from the source at the vessel to the sea bottom and return. When mounted to the vessel, it is called a fathometer. Typical low frequency equipment operates at 12 kHz and high frequency equipment at 200 kHz. The major difference between various types of echosounders is the frequency. Transducers can be classified according to their beam width, frequency, and power rating. Beam width is determined by the frequency of the pulse and the size of the transducer. In general, lower frequencies produce a wider beam, and at a given frequency, a smaller transducer would produce a wider beam. Lower frequencies penetrate deeper into the water, but have less resolution at depth. Higher frequencies have a greater resolution in depth, but less range.

Remote sensing equipment will be used in some of the activities that are part of the proposed action, such as the location of suspected MEC/MPPEH and underwater investigations. This equipment includes side scan sonar, which the Navy notes will be operated in a frequency range of 400 to 1600 kHz (CH2M Hill 2018). Surveys are conducted over several days to weeks along transects in different locations within UXO 16 so exposure to sound from these surveys is temporary. Remote sensing equipment is also used to reacquire the location of suspected MEC/MPPEH as part of removal activities. The use of equipment for this purpose is even shorter term than for surveys.

Fin and right whales were found to react to frequencies from 15 Hz to 28 kHz, but not to frequencies above 36 kHz (Watkins 1986). These and other toothed whales, including sperm whales, are considered mid-frequency cetaceans with a generalized hearing range from 150 Hz to 160 kHz (Southall et al. 2007b; NMFS 2016). ESA-listed sea turtles are not expected to detect signals emitted by navigational equipment, as the operating frequency range is well outside the hearing range of sea turtles, which appears to be 100 to 400 Hz. Most fish species can hear sounds between 50 and 1,000 Hertz (Hz) with most ESA-listed fish studied (largely salmonids and sturgeon) having a hearing range below 400 Hz so fish without hearing specialization are not expected to detect signals emitted by navigational and survey equipment. Therefore, we believe the effects of sound from vessel noise, navigation equipment, and survey equipment operated in a frequency range of 400 to 1600 kHz on sperm whales, green, leatherback and hawksbill sea turtles, and Nassau grouper will be insignificant and thus not likely to adversely affect these animals.

Coring equipment could be used to collect coral tissue samples. Hydraulic or pneumatic drills are often used to collect cores (Weinzierl, et al. 2016), typically with diameters up to 4-in, from large coral colonies. Due to the size of these drills, noise produced by coring is not comparable to hydraulic drills used in underwater construction for which source levels of 164.2 to 179.2 dB re: 1 µPa at 1 m (root mean square [rms]) are reported for sound pressure levels examined at frequency bands of 50-1000 Hz and 100-400 Hz (Reine and Clarke 2014). Coral coring is done quickly and the sound produced is not expected to result in levels above those produced by other equipment discussed in this section or to be significantly above ambient noise levels recorded in reef environments. For example, snapping shrimp generate sounds with the most energy at frequencies of 2-5 kHz and individual snaps can have peak-to-peak pressure source levels up to 189 dB at 1 m (Au 1998). The Navy indicated that sampling will be limited to a small number of coral colonies (50 in total over the consultation lifetime), if it occurs at all, and will take place only in areas with large coral colonies, meaning that sperm whales are not expected to be present in areas where coral tissue samples collected via coring would occur. Therefore, no effects from sound produced by coral coring are expected to sperm whales. We believe the effects to green, leatherback and hawksbill sea turtles, and Nassau grouper from the noise associated with coral coring will be insignificant and thus not likely to adversely affect these animals.

The installation of in-water structures such as buoys, floating barriers, and associated anchor systems, will also result in temporary impacts associated with noise generated by coring and drilling equipment used to bore holes in hard substrate to install anchor pins and jacks used to install Manta Ray™ anchors in sand and other bottom substrates. Manta Ray™ anchors are typically installed using a hydraulic jack. Anchor pins are installed using a hydraulic drill or corer with a diameter up to 4-in. The equipment used to install anchor pins may be the same as that used to collect coral cores and the noise generated during installation of anchor pins is expected to be similar. Hydraulic jacks used to push Manta Ray™ anchors into the sediment may generate more noise than drills used to install anchor pins and the noise may last up to an hour, depending on the depth to which the anchors are being installed. However, none of the sound produced by the installation of in-water structures will be of long duration and the frequencies and source levels are not expected to cause anything other than temporary disturbance of animals, including green, leatherback and hawksbill sea turtles, and Nassau grouper. None of the structures are expected to be installed in deeper waters so sperm whales will not be affected by noise associated with the installation of in-water structures. We believe the effects to green and hawksbill sea turtles and Nassau grouper from the noise associated with the installation of in-water structures, including drilling and coring and the use of a hydraulic jack, will be insignificant and thus not likely to adversely affect these animals.

If new equipment for remote sensing or other activities associated with the proposed action is proposed in the future that will operate at different frequencies and have different source levels, the potential effects of the use of this equipment on ESA-listed species would have to be analyzed as part of a step-down consultation as described in Section 3.3.2 and may require reinitiation of consultation depending on the potential effects of the equipment on ESA-listed species.

8.1.5 Entanglement

The offshore anchorage areas are the only locations within UXO 16 where activities will potentially overlap with sperm whales. Activities such as investigations using towed equipment and removal activities requiring remote lifting and tow of suspected munitions items could pose an entanglement risk to sperm whales in these areas, but the fact that observers will be on board the vessels and lines used for towing float makes it unlikely that sperm whales will become entangled. Several of the activities that are part of the proposed action will result in lines in the water that could pose an entanglement risk for green and hawksbill sea turtles. However, based on a review of unpublished sea turtle stranding data for Puerto Rico from 1989-2009 (PRDNER), dead sea turtles have been found entangled in fishing nets and line and abandoned cargo nets, but no strandings were attributable to entanglement in buoy and in-water structure anchor lines. Fish entanglement in tackle associated with in-water structures have not been reported and towing, which takes place near the water surface, is not expected to result in encounters with fish, which will swim away from the disturbance. Therefore, we believe there will be no effect of entanglement in tow lines or lines associated with in water structures to Nassau grouper. We believe the effects to sperm whales and green, leatherback and hawksbill sea turtles from entanglement in towlines and lines associated with in-water structures will be discountable and thus not likely to adversely affect these animals.

Entanglement in lines associated with towed equipment and in-water structures could result in breakage and abrasion of ESA-listed coral colonies. However, slack in towlines only occurs when the tow vessel is not underway and would not occur in shallow water areas containing corals based on the SOPs developed by the Navy. In addition, entanglement of towlines would potentially lead to damage of the equipment being towed. Entanglement of towlines used to move MEC/MPPEH suspected to present a detonation hazard would jeopardize worker safety. For these reasons, the SOPs developed by the Navy include the use of observers and the use of procedures to minimize slack on the towline. Similarly, for in-water structures, the design of the joins between parts of the structures, such as floating waterway barriers, is meant to reduce slack in the connections while still allowing movement with waves and currents. These structures and their associated anchor tackle are sited to minimize potential impacts to areas containing ESA-listed corals associated with swing on the anchor and slack in the lines. The anchors for in-water structures are also designed with back-up anchors to minimize the potential for structures and anchors to come lose during storms. Therefore, we believe the effects of entanglement on ESA-listed corals will be discountable and thus not likely to result in adverse effects.

Entrapment is not expected to affect sperm whales, adult sea turtles, or Nassau grouper due to the type of in-water structures the Navy anticipates installing in waters within UXO 16, most of which are on the surface or do not form enclosed spaces where animals could be trapped.

The effects of entrapment of hatchling sea turtles in in-water structures such as floating barriers are discussed in Section 8.2.

8.1.6 Sediment Resuspension

All the activities associated with the proposed action that have the potential to disturb the bottom, including removal of MEC/MPPEH from the marine bottom, sediment sampling, installation of in-water structures, underwater investigations requiring excavations, and associated vessel anchoring during operations. Bottom disturbance is expected to cause sediment resuspension and transport. However, because sand bottoms interspersed with seagrass beds and coral habitats characterize the majority of areas within UXO 16 where activities will disturb bottom sediments, sediment resuspension and transport is expected to be minimal because of the large grain size and weight of sand, which lead to sand resettling to the bottom quickly after a disturbance. In addition, areas with seagrass will have little sediment resuspension and transport because the seagrass serves as a natural sediment trap, unless large areas are excavated, which is not expected to be required as part of the proposed action unless large bombs are found near the surface in areas with seagrass beds. Similarly, disturbance in coral habitats will not generate large amounts of sediment because coral habitats are not characterized by high sediment content and excavation of coral habitat is not expected as part of the proposed action. The activities associated with the proposed action are expected to be completed over the course of several days to weeks in different sites within UXO 16, with work occurring only during daylight hours. Any sediment resuspension and transport would be temporary. Sediment cores associated with sediment sampling for munitions constituents will be done by hand using a collection tube in uncolonized bottom substrate and are not expected to lead to any sediment resuspension. Therefore, we believe the effects to sperm whales, green, leatherback and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat due to sediment resuspension and transport will be insignificant and thus not likely to result in adverse effects to these species or habitat.

If changes are proposed in the future that would result in large excavations of seagrass and/or coral areas with the potential to generate measurable concentrations of sediment in the water column for longer periods of time, reinitiation of consultation may be required.

8.1.7 Habitat Loss or Damage

Green and hawksbill sea turtle refuge and foraging habitat and habitat used by Nassau grouper could be lost because of the installation and operation of in-water structures. The 3-ton bulk cement anchors that may be used to anchor in-water structures have a footprint of 64 ft^2 and may have a single helical or Manta Ray™ anchor as a secondary anchor. Helical anchors have a footprint of 78 in^2 and Manta Ray of 60 in^2. All three of these anchor types would be used in unconsolidated bottom substrates. Pin anchors have a footprint of 28 in^2 and would be used in hard bottom habitats. The ARMS will be installed in sand bottom, hardbottom with turf algae, and rubble bottom and thus are not expected to impact refuge and foraging habitat used by green and hawksbill sea turtles or Nassau grouper. Given the existing acreage of seagrass and coral habitats around Vieques, estimated as 3,557 acres and 5,198 acres, respectively based on the WAA (CH2M Hill 2018), we believe the effects to green and hawksbill sea turtles and Nassau grouper as a result of habitat impacts from the installation of different anchor systems for in-water structures in UXO 16 will be insignificant and thus not likely to result in adverse effects to these species. As stated previously, leatherback sea turtles do not utilize water in the action area other than for transit during nesting season; therefore, any impacts to benthic habitat would not have an effect on this species of sea turtle.

Green and hawksbill sea turtle and Nassau grouper habitat could also be damaged as a result of the removal of MEC/MPPEH and underwater investigations that require excavation in seagrass and unconsolidated bottom or removal of items from the surface in coral habitats, including items that may have benthic organisms such as sponges that may be eaten by hawksbill sea turtles growing on them. There are thousands of potential MEC/MPPEH items within UXO 16, many of which are in areas containing seagrass beds and coral habitats that may be used by green and hawksbill sea turtles and Nassau grouper. Many of the items may be on the surface, in which case habitat disturbance in minimal and the only impacts may be the loss of organisms that could serve as prey species encrusted on removed items. Other items may be below the surface, in which case excavation of unconsolidated substrate, including in areas of seagrass beds, would result in disturbance of habitat used by green sea turtles and Nassau grouper in particularly as well as juvenile hawksbill sea turtles. However, the Navy has developed SOPs to minimize this disturbance, including cutting and folding back seagrass and then replanting it over the disturbed area once items have been excavated. Based on underwater cleanup done to date in UXO 16 and similar work being done by the USACE around Culebra and its surrounding islands and cays, which is a Formerly Used Defense Site, habitat disturbance from the removal of multiple munitions items is minimal, even for larger items such as large bombs. Similarly, the footprint of bottom-operated equipment to locate suspected MEC/MPPEH, which is only used in areas with unconsolidated bottom, is extremely small in comparison to the habitat areas available to sea turtles and Nassau grouper within UXO 16. The footprint of tripods that are mounted around suspected MEC/MPPEH items thought to pose an explosive hazard for remote lifting and towing is very small in comparison to the area of available habitat within UXO 16. If removal methods were to include BIPs or encapsulation, which could generate larger areas of habitat impacts, there would still be thousands of acres of habitat available for use by green and hawksbill sea turtles and Nassau grouper. Therefore, we believe the effects to green and hawksbill sea turtles and Nassau grouper because of habitat impacts from the investigation and removal of suspected MEC/MPPEH in UXO 16 will be insignificant and thus not likely to result in adverse effects to these species.

Habitat loss or damage to elkhorn and staghorn coral critical habitat associated with the installation of anchor pins and removal of suspected MEC/MPPEH is discussed in Section 8.2.

8.1.8 Bycatch

Bycatch of juvenile green and hawksbill sea turtles and Nassau grouper during biological sampling events as part of the proposed action is discussed in Section 8.2.

Cast nets and fish traps that will be used for biological sampling will not result in bycatch of sperm whales given the locations where this fishing gear is likely to be deployed (in nearshore shallow waters) and the type and size of gear in comparison to the size of sperm whales.

8.1.9 Organism Collection and Transplanting

Organism collection and transplanting will occur because of biological sampling and during removal of MEC/MPPEH from UXO 16. Biological sampling using cast nets and traps and corers to collect coral tissue samples will have no effect on sperm whales or adult green and hawksbill sea turtles. Biological sampling using fishing gear and the collection of coral tissue samples will affect juvenile green and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat and these effects are discussed in Section 8.2.

The collection and transplanting of corals, seagrass, and other sessile benthic organisms from areas where the removal of munitions items occurs or growing on the items to be removed will result in some loss of or damage to prey items and habitat used by green and hawksbill sea turtles and Nassau grouper, as also discussed in Section 8.1.7. The Navy has developed SOPS to insure that impacts to seagrass from removal activities are minimized, including through replanting of seagrass in situ and transplanting seagrass to other areas when necessary and feasible. The Navy has also developed SOPs to minimize the loss and degradation of coral habitats associated with removal activities. Therefore, we believe the effects to green, leatherback and hawksbill sea turtles, and Nassau grouper from the collection and transplant of organisms associated with removal activities will be insignificant and thus not likely to adversely affect these animals. The effects of collection and transport on ESA-listed corals and elkhorn and staghorn coral critical habitat are discussed in Section 8.2.

8.1.10 **Metal Leachate and Contaminant Release**

Sacrificial anodes are used on vessel motors and on some components of in-water structures to reduce metal degradation associated with exposure to saltwater. Zinc anodes are often used in saltwater, but aluminum is now commonly used instead of zinc because it is lighter and can last up to 50 percent longer than zinc in saltwater. Sacrificial anodes help preserve other metals that make up the propeller and propeller shaft or tackle components that are part of the anchor system of in-water structures because they are more electrically active than the other metals. When electrically connected to them in saltwater, the metal from the sacrificial anode becomes the material that gives up electrons and dissolves rather than the metals in boat motor parts and anchor systems. The Navy did not specify which anode material will be used on boat motors and to protect anchor system components or other parts of in-water structures so we analyze the potential effects of both zinc and aluminum to sperm whales, green, leatherback and hawksbill sea turtles, Nassau grouper, and ESA-listed corals.

In 1986, an analysis of water and sediment samples identified cadmium, copper, lead, mercury, nickel, selenium, and zine at nine stations in the USVI. A 1992 evaluation of aquatic use impairments in the Caribbean reported that arsenic, cadmium, chromium, copper, lead, zinc, aluminum, and mercury were detected at low levels suspended sediment samples analyzed between 1972 and 1979. One hundred and eighty five chemical contaminants were analyzed in sediments, and a series of sediment toxicity bioassays were conducted along with a characterization of the benthic infaunal community to assess the presence and effects of chemical contaminant stressors in the St. Thomas East End Reserves. Zinc, copper, lead, and mercury were detected above a NOAA sediment quality effects-range low guideline at one or more sites, indicating effects were possible (Pait et al. 2016). Coral (*Porites astreoides*) and conch (*Lobatus gigas*) were among the species evaluated for contaminants in tissues. Contaminants found in coral were similar to the concentration ranges reported in corals from other reef areas in the U.S. Caribbean while conch had lower contaminant body burdens relative to published data from south Florida and some other areas of the Caribbean.

Elevated aluminum levels in acidic waters may be toxic to fish, with different effects depending on life history stage. At low pH levels (4.2 to 4.8), the presence of aluminum (up to 0.2 milligrams per liter [mg/L] for white suckers; 0.5 mg/L for brook trout) was beneficial to egg survival (Baker and Schofield 1982). On the other hand, concentrations of 0.1 mg/L for white suckers and 0.2 mg/L for brook trout resulted in measurable reductions in survival and growth of larvae and postlarvae at all pH levels studied (4.2 to 5.6) with toxicity being greatest at pH levels of 5.2 to 5.6 (Baker and Schofield 1982). At moderate acidity (pH 5.5 to 7.0), fish and invertebrates may be stressed due to aluminum adsorption to gill surfaces and subsequent asphyxiation (Sparling et al. 1997). Marine waters in the action area are expected to have a pH above 7 so the effects reported for acidic waters are not expected to occur in the action area. Concentrations greater than 1000 mg/kilogram in food may be toxic to mammals in terrestrial environments (Sparling et al. 1997), but this may not be true for marine mammals. EPA found there were insufficient data to recommend water quality criteria for aluminum for estuarine/marine waters in 2018 (EPA 2018). New acute toxicity data are now available for five families representing five species of estuarine/marine organisms, which was not the case in 1988 (EPA 2018). The most sensitive species was a polychaete worm (*Ctenodrilus serratus*) with a Species Mean Acute Value (SMAV) of 97.15 micrograms per liter (µg/L) total aluminum. The most tolerant species was a copepod (*Nitokra spinipes*) with a SMAV of 10,000 µg/L (EPA 2018). No acute tests on estuarine/marine fish species meeting the requirements for EPA to use in establishing water quality criteria were available (EPA 2018). Leachate of aluminum from sacrificial anodes is not expected to be present in concentrations similar to those reported in toxicity studies.

Zinc is an essential trace mineral that is toxic in excess amounts. A factorial study exploring the effects of exposure time and concentration on the toxicity of zinc to rainbow trout reported no mortality in the first four hours of zinc exposures ranging from 5,000 to 13,000 µg/L (Gündoğdu 2008). For fish, the lethal concentration at which 50 percent of organisms suffered mortality (LC50) for exposures to zinc ranged from 25.54 to 170,280 µg/L (n=71), representing 28 species from 24 studies. Fish chronic exposure-response thresholds for exposures to zinc ranged from 95.55 to 9,460 µg/L (n=13), representing seven species from nine studies. Invertebrate LC50s for exposures to zinc ranged from 7.38 to 1,655,000 µg/L (n=367), representing 109 species from 91 studies. Algae, which can be considered a surrogate for coral zooxanthellae, chronic exposure-response thresholds for exposures to zinc ranged from 17.97 to 21,758 µg/L (n=58), representing 17 species from 22 studies. As for aluminum, leachate of zinc from sacrificial anodes is not expected to be present in concentrations similar to those reported in toxicity studies.

A study by Bird et al. (1996) found that sacrificial anodes can cause measurable increases in concentrations of dissolved zinc in marinas, particularly in enclosed marinas and sediments in these areas, and to a lesser extent in nearby estuaries. The local increase in concentrations of dissolved zine near open marinas may be between 2 and 5 µg/L while concentrations in sediments within marinas may be up to twice background levels (Bird et al. 1996). Aluminum anodes were found to release aluminum to sediments that is then partly bound to the acid-soluble fraction of sediment. The authors theorize that this is due to the integration of aluminum released from the anodes into the calcareous deposits that form at the anode surface, which tends to flake off and be mixed into the sediment (Leleyter et al. 2018). However, the increase in total aluminum content in sediments is only 5 percent from natural levels in the case of sacrificial anodes and the analysis of total content cannot discriminate the aluminum from anthropogenic source because it is negligible compared to the natural amount present in sediments (Leleyter et al. 2018).

As part of the proposed action, there will be no concentration of vessels in a particular area for extended periods, meaning metals from sacrificial anodes associated with vessel operation associated with the proposed action are not expected to be detectable in waters and sediments in UXO 16. While sacrificial anodes associated with in-water structures do remain in the same area for extended periods of time, the size (several inches) and number (one per U-bolt on concrete anchors for Bahia Icacos waterway barrier, as an example) of these compared with the size of the area where the structures are located means that metals from these anodes are not expected to be measurable. Therefore, we believe the effects of zinc or aluminum leaching from sacrificial anodes on sperm whales, green, leatherback and hawksbill sea turtles, Nassau grouper, ESA-listed coral, and elkhorn and staghorn coral critical habitat will be insignificant and thus not likely to adversely affect these species and critical habitat.

Munitions compounds may leach from underwater MEC/MPPEH into the water column and sediments, be released due to breakage or spillage during removal activities, or be present in sediments that are resuspended during underwater investigation and removal activities. Coral cell toxicity assays were conducted to test three nitrotoluene munitions compounds: TNT and two of its major breakdown products (2,4-DNT and 2,6-DNT); two nitroamines: RDX and HMX; and one nitrophenol: picric acid (2,4,6-trinitrophenol). Woodley and Downs (2014) found picric acid to be the most toxic overall with the lowest LC_{50} (concentration of the compound that is lethal for 50 percent of the exposed population) at 10.5 µg/L for *Pocillopora damicornis* calicoblast cells (which are involved in the production of the coral skeleton). On the other hand, 2,4,6-TNT was found to be the most toxic for gastrodermal cells (which form the lining of the gastrovascular cavity) of this coral and an LC_{50} could not be determined for RDX or HMX under any of the laboratory conditions (Woodley and Downs 2014). The sensitivity of coral cells to TNT was also found to be more pronounced in the presence of light versus in dark conditions (Woodley and Downs 2014). Woodley and Downs (2014) also tested calicoblast and gastrodermal cells from three species of corals to determine whether there was a between-species difference in sensitivity to munitions compounds, specifically 2,6-DNT, which was used in the laboratory tests. There was a marked difference between species in terms of sensitivity though the gastrodermal cells of all three species were found to be more sensitive than the calicoblast cells by orders of magnitude. *Pocillopora damicornis* was more sensitive than *Porites divaricata* and *Porites lobata* with an LC_{50} for gastrodermal cells of 1,844 µg/L (Woodley and Downs 2014).

Woodley and Downs (2014) also tested the toxicity of munitions compounds of *Symbiodinium* sp. (species that are coral zooxanthellae) and found TNT was the most toxic of the nitrotoluenes with an EC_{50} (effects concentration at which 50 percent of the organisms show an adverse response) of 544 µg/L for cell growth (2,4,6-TNT) and an EC_{50} of 2,810 µg/L for photosynthetic efficiency (2,3-DNT). Coral fragments were also used to conduct exposure/response studies using 96-hour exposures to three munitions compounds, RDX, 2,3-DNT, and TNT. Woodley and Downs (2014) found signs of lethal toxicity in *Pocillopora damicornis* of 2,3-DNT at concentrations of 2,000 µg/L and higher within 18 hours of exposure and sublethal effects at 292 µg/L. Woodley and Downs (2014) also found TNT showed toxic effects in *Porites divaricata* fragments with changes in polyp behavior and tissue integrity, and necrosis at concentrations of 100 µg/L and higher. The concentrations at which toxic effects of munitions compounds were observed in the Woodley and Downs (2014) laboratory experiments are not likely to be representative of the concentrations at which compounds are present in the environment within UXO 16. Whitall et al. (2016) sampled queen conch from three sites around Vieques that are within UXO 16 for metals, pesticides, and energetic compounds associated with munitions and found that concentrations of pollutants were within the range of values reported in other studies in the Caribbean where military practices have not occurred. Munitions compounds were not detected in any samples.

Similarly, a longitudinal study to compare lead, cadmium, and copper content in manatee grass (*Syringodium filiforme*) did not find a significant difference between the bioaccumulated concentration of these metals at the bombing range off Vieques versus the reference site used in the study (Díaz et al. 2018). Environmental samples typically show that concentrations of munitions compounds in water and sediment in sites contaminated with military debris are generally very low, meaning ecological risk is thought to be low (Beck et al. 2018). However, there could be sublethal genetic and metabolic effects for organisms with chronic exposure to these compounds (Beck et al. 2018). For Vieques, a proof-of-concept study to evaluate the ecological risk from exposure to munitions compounds was conducted using grab sampling and Polar Organic Chemical Integrative Samplers (passive sampling devices). The concentrations detected by the passive samplers were 10 to 1,000,000 times lower than hazardous concentrations to five percent of species (HC_5) generated from the most up-to-date and comprehensive species sensitivity distributions (Rosen et al. 2017). Similarly, an assessment of chemical contamination in Bahia Salinas del Sur found only one of six coral samples collected from the stern of the USN*Killen* (a vessel that served as a target during live-fire military exercises) contained detectable residues of TNT, 252 µg/g TNT (Porter et al. 2011). Seawater samples were found to contain high levels of TNT within one cm of a submerged bomb but the concentrations of TNT and other munitions compounds were orders of magnitude lower within 10 cm of the bomb and the concentrations of munitions compounds in sediment samples showed a similar decline to no detection 2 m from the bomb (Porter et al. 2011). Therefore, while it is possible that corals growing on a munitions item could demonstrate sublethal responses such as declines in growth, concentrations of munitions compounds leaking from munitions items would have to be at or above those found by Woodley and Downs (2014) to cause sublethal effects to corals and their zooxanthellae. None of the studies of organisms or chemical concentrations of compounds in organisms, the water column, or sediments in UXO 16 (Díaz et al. 2018; Whitall et al. 2016; Rosen et al. 2017; Porter et al. 2011), other than those by Porter et al. (2011) at a submerged bomb were close to these concentrations. Therefore, we believe the effects of munitions compounds released to the water column or in resuspended sediments during investigation and removal activities on sperm whales, green, leatherback and hawksbill sea turtles, and Nassau grouper will be insignificant and thus not likely to adversely affect these species. We believe the effects of munitions compounds on ESA-listed corals will be discountable and therefore not likely to adversely affect these species. Munitions compounds will not affect elkhorn and staghorn coral critical habitat.

8.2 Exposure, Response, and Risk Analyses

In the previous sections, we described the stressors resulting from the action and determined that nonintentional detonation and BIPs are likely to adversely affect ESA-listed sperm whales; green, leatherback and hawksbill sea turtles; Nassau grouper, and corals. We also determined that habitat loss or damage from nonintentional detonation and BIPs, removal of items from coral habitats when those items are cemented into the hard substrate, encapsulation of items in reefs, and anchor pin installation in hard bottom habitats are likely to adversely affect ESA-listed corals and elkhorn and staghorn coral critical habitat. Collisions of munitions items and towed equipment are likely to adversely affect ESA-listed corals. Bycatch as part of biological sampling activities is likely to adversely affect Nassau grouper and smaller size classes of green and hawksbill sea turtles. Organism collection and transport is likely to adversely affect ESA-listed corals and Nassau grouper. Entrapment associated with the installation and operation of in-water structures such as floating barriers is likely to adversely affect hatchling green, leatherback, and hawksbill sea turtles.

In the following section, we consider the exposures that could cause an effect on ESA-listed species that are likely to co-occur with the effects of the stressors identified in the previous paragraph on the environment in space and time, and identify the nature of that co-occurrence. We consider the frequency and intensity of exposures that could cause an effect on ESA-listed species and, as possible, the number, age or life stage, and gender of the individuals likely to be exposed to the action’s effects and the population(s) or subpopulation(s) those individuals represent. We also consider the responses of ESA-listed species to exposures and the potential reduction in fitness associated with these responses.

As discussed in Section 3, existing boat access ramps within the action area will be used to launch and recover vessels used to conduct the activities described in this Opinion. If additional access ramps, whether temporary or permanent, are required in order to conduct any of the activities that are part of the proposed action, a step-down consultation will be required in order to determine whether the effects of construction and operation of the access ramp(s) on sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat will differ from the effects analyzed in this Opinion, whether additional PDCs are needed, and whether the ITS needs modification to address additional incidental take.

8.2.1 Definition of Take, Harm, and Harass

Section 3 of the ESA defines take as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. We categorize two forms of take, lethal and sublethal take. Lethal take is expected to result in immediate, imminent, or delayed but likely mortality. Sublethal take is when effects of the action are below the level expected to cause death, but are still expected to cause injury, harm, or harassment. Harm, as defined by regulation (50 C.F.R. §222.102), includes acts that actually kill or injure wildlife and acts that may cause significant habitat modification or degradation that actually kill or injure fish or wildlife by significantly impairing essential behavioral patterns, including, breeding, spawning, rearing, migrating, feeding, or sheltering. Thus, for sublethal take we are concerned with harm that does not result in mortality but is still likely to injure an animal.

NMFS has not defined "harass" under the ESA by regulation. However, on October 21, 2016, NMFS issued interim guidance on the term "harass," defining it as to "create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering." For this consultation, we rely on this definition of harass when assessing effects to all ESA-listed species except marine mammals.

Our October 21, 2016, guidance states that our "interim ESA harass interpretation does not specifically equate to MMPA Level A or Level B harassment, but shares some similarities with both levels in the use of the terms 'injury/injure' and a focus on a disruption of behavior patterns. NMFS has not defined 'injure' for purposes of interpreting Level A and Level B harassment but in practice has applied a physical test for Level A harassment." Under the MMPA, harassment is defined as any act of pursuit, torment, or annoyance which:

- Has the potential to injure a marine mammal or marine mammal stock in the wild (Level A Harassment); or
- Has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering (Level B Harassment).

8.2.2 Exposure to Stressors

Sperm whales in the Caribbean are considered a distinct stock and are present largely during their winter migration through the warmer waters of the Caribbean. Female sperm whales may give birth in waters off Vieques, based on observations of mother-calf pairs in the action area during past surveys (Roden and Mullin 2000; GMI 2001; Mignucci-Giannoni et al. 2000).

Juvenile sperm whales has also been observed in deeper waters of the action area during past surveys (Roden and Mullin 2000; GMI 2001) and an immature sperm whale stranded on Vieques in 2013. Therefore, depending on the time of year when removal activities are occurring, juveniles and mother-calf pairs and juveniles may be present in the action area, though likely in low numbers given that a maximum of two animals at one time (a mother-calf pair) were sighted during past surveys (Roden and Mullin 2000; GMI 2001; Mignucci-Giannoni et al. 2000).

From 1992 to 2000, 151 green sea turtle nests were deposited on nesting beaches on Vieques Island. Nesting was largely every two years with numbers ranging from 19 in 1992 to 54 in 2000 (PRDNER nesting data; Matos et al. 1992; Belardo and Matos 1993; Belardo et al. 1994; Belardo et al. 1995; Belardo et al. 1996; Belardo et al. 1997; Belardo et al. 1998;1999;2000;2001). In 2019 (up to September 6), USFWS observed a total of 140 green sea turtle nests on beaches in the VNWR public areas, DNER reserve, and in the VNWR restricted area. Green sea turtles are also frequently sighted in waters around Vieques. It is not possible to determine which DPS (North or South Atlantic) animals belong to, particularly given that Puerto Rico is on the border between these two DPSs and Vieques is geographically considered part of the "Virgin Islands." Based on calculations of the potential number of adult, juvenile, and hatchling green sea turtles (see Section 7.1.2), we estimate that 106 adults, 1,751 juveniles, and up to 4,788 hatchlings could be exposed to removal, in-water structure installation and operation, and biological sampling activities.

In 2019 (up to September 6), USFWS observed a total of 69 leatherback sea turtle nests on beaches in the VNWR public areas, the DNER reserve, and the VNWR restricted area. This species is only present on beaches and in waters of Vieques during its nesting season, which peaks from April – July. Leatherback sea turtles are pelagic and only enter nearshore waters during their nesting season. Based on calculations of the potential number of adult and hatchling leatherback sea turtles (see Section 7.1.2), we estimate that 30 adults and up to 2,295 hatchlings could be exposed to stressors depending on when removal activities and installation and operation of in-water structures occur.

From 1991 to 2000, 285 hawksbill sea turtle nests have been deposited on nesting beaches on Vieques Island. Nesting was annual and occasionally year-round with peaks on Tamarindo Sur (44), Fanduca Beach (46), Jalova Beach (44), and Jalovita Beach (45) over this period (PRDNER nesting data; Matos et al. 1992; Belardo and Matos 1993; Belardo et al. 1994; Belardo et al. 1995; Belardo et al. 1996; Belardo et al. 1997; Belardo et al. 1998;1999;2000;2001). Annual numbers of nests ranged from 24 in 1991 to 50 in 2000. In 2019 (up to September 6), USFWS observed a total of 58 hawksbill sea turtle nests on beaches in the VNWR public areas, the DNER reserve, and the VNWR restricted area. Hawksbill sea turtles are also frequently sighted in waters around Vieques. Based on calculations of the potential number of adult, juvenile, and hatchling hawksbill sea turtles (see Section 7.1.2), we estimate that 48 adults, between 963 and 4,377 juveniles, and up to 3,898 hatchlings could be exposed to removal, in-water structure installation and operation, and biological sampling activities.

Nassau grouper were observed in the action area in the past during in-water surveys in coral reefs, seagrass beds, and colonized hard bottom (Department of the Navy 1979;1986; GMI 2003; García-Sais et al. 2001; García-Sais et al. 2004). Seagrass and coral habitats in nearshore waters provide nursery habitat for juvenile Nassau grouper while adults use reef habitats, usually in deeper waters. Historic Nassau grouper SPAGS include the eastern point of Vieques (Ojeda-Serrano et al. 2007). During more recent surveys by NCCOS around Vieques, this species was not observed, but similar large grouper species were observed in up to 1 percent of fish count surveys (Bauer and Kendall 2010). However, surveys of SPAGS in the USVI and the west coast of Puerto Rico have reported multispecies aggregations that have included some Nassau grouper (Kadison et al. 2009; Schärer et al. 2009) and other surveys in UXO 16 reported Nassau grouper observations that were not quantified because the observations did not occur during the actual point count surveys (GMI 2003). Thus, it is likely that juvenile and adult Nassau grouper may be present in the action area in low numbers during removal and biological sampling activities.

There are hard bottom and reef habitats containing coral colonies of ESA-listed corals in waters around Vieques based on the WAA and other surveys done by the Navy and its contractors and previous surveys (see Section 7.1.4). Lobed star coral appears to be the dominant live coral species on reef and hard bottom habitats around Vieques (Bauer and Kendall 2010) with mountainous star corals also being common. Staghorn coral was observed in 12 percent of NCCOS surveys (Bauer and Kendall 2010). Elkhorn coral is most common in shallower depths (up to 5 m) and has been observed during site inspections associated with underwater work in UXO 16 including off Cayo la Chiva, in Bahia Icacos, and in-water portions of SWMU 4. Pillar coral, boulder star coral, mountainous star coral, and rough cactus coral have been reported in studies conducted in UXO 16 in 2001-2005 (Bauer et al. 2008). CH2M Hill (2018) estimated that there could be up to 5,173 ESA-listed coral colonies affected by the activities that are part of the proposed action, specifically because of the location of these corals adjacent to or growing on suspected MEC/MPPEH. We can use the percentage of ESA-listed corals of each species present in various surveys conducted in the action area (NCRMP; Bauer and Kendall 2010) to calculate approximately how many colonies of elkhorn, staghorn, pillar, rough cactus, lobed star, boulder star, and mountainous star may comprise this estimate. We would therefore assume that elkhorn corals make up one percent of ESA-listed corals, staghorn 12 percent, pillar coral 0.28 percent, lobed star coral 0.5 percent, boulder star coral 20 percent, and mountainous star coral 10 percent. Because both pillar and rough cactus coral are rare, we assume that rough cactus coral makes up the same percentage as pillar coral. Using these percentages, we determined that, of the 5,173 ESA-listed coral colonies the Navy estimates may be affected by the proposed action, 52 could be elkhorn coral colonies, 621 could be staghorn coral colonies, 14 could be pillar coral colonies, 14 could be rough cactus coral colonies, 26 could be lobed star coral colonies, 1,035 could be boulder star coral colonies, and 517 could be mountainous star coral colonies. However, because there may be significant variability between sites containing ESA-listed corals, including due to differences in water depths, and most of the NCRMP stations around Vieques are in waters around 60 ft in depth, these estimates may not be the most accurate characterization of the numbers of colonies of each ESA-listed coral species that will be exposed to stressors from the action that are likely to result in take of colonies.

Elkhorn and staghorn coral critical habitat is present in the action area. The Navy determined that 5,198.2 acres within UXO 16, or 48.9 percent, of the benthic habitat contains coral reef and hard bottom (see Section 7.1.5). Of this acreage, hard bottom in the form of rock/boulder, pavement, and pavement with sand channels comprises 1,940.7 acres and reef in the form of aggregate reef, individual patch reef, aggregated patch reefs, and spur and groove comprises 800.8 acres. Although the WAA encompassed deeper areas such as the offshore anchorages, most of the areas identified as containing coral habitat are within the 30 m depth and may contain the PBF for elkhorn and staghorn coral critical habitat.

8.2.3 **Response**

Given the exposure discussed above, in this section we describe the range of responses among ESA-listed sperm whales; green (North and South Atlantic DPSs), leatherback, and hawksbill sea turtles; Nassau grouper; corals, and elkhorn and staghorn coral critical habitat, as applicable, associated with equipment collisions, underwater detonations, habitat loss or damage, bycatch, and organism collection and transport associated with activities that will be implemented as part of the action. For the purposes of this consultation, our assessment tries to detect potential lethal, sub-lethal (or physiological), and behavioral responses that might reduce the fitness of individuals.

8.2.3.1 Equipment Collisions

Equipment collisions with ESA-listed corals may occur. A collision with ESA-listed coral colonies was reported as part of past survey work conducted in UXO 16 (B. Doerr, Jacobs [formerly CH2MHill], personal communication to L. Carrubba, NMFS, April 21, 2016). The collision resulted in breakage of two coral colonies. The equipment was modified to minimize the potential for additional collisions and no further interactions with ESA-listed coral colonies were reported during the survey. The SOPs for the use of towed equipment, as well as for towing munitions items that have to be moved remotely and for operating vessels, that were incorporated in the PDCs and some additional PDCs in Section 3.3.2, were developed by the Navy in order to minimize interactions between vessels and equipment used for surveying and removal activities. While implementation of the PDCs is expected to minimize the potential for collisions with ESA-listed coral colonies, the potential cannot be eliminated, particularly when vessels are towing equipment or MEC/MPPEH items for transport to a disposal area. Collisions with ESA-listed corals would cause breakage and abrasion of the coral colonies. In addition, colonies affected by breakage or abrasion, which leads to exposed tissue, are more susceptible to bleaching and disease. Collisions with ESA-listed corals during periods of elevated sea surface temperatures and/or disease outbreaks would increase the likelihood that colonies affected by the collisions will bleach and/or be infected by disease. Depending on the size of the colony, the size of the equipment, and the severity of the collision, the colony could be killed by the impact. Fragmented colonies could survive and the fragments could also regrow but reproduction would not occur for one to two years following the collision as the corals would be dedicating resources to regrowth rather than reproduction. Therefore, there could be fitness consequences to a small number of ESA-listed coral colonies (based on information from surveys conducted to date during which only two coral colonies were damaged by collision) associated with equipment collisions. The effects of collisions with ESA-listed corals by equipment are discussed further in Section 8.2.4.

8.2.3.2 Underwater Detonations

Non-intentional detonations, which may occur when a munitions item explodes as a result of movement associated with removal actions, and BIPs to detonate items presenting an explosive hazard, could result in noise levels that cause adverse effects to sperm whales, Nassau grouper, leatherback, green, and hawksbill sea turtles, ESA-listed corals, and elkhorn and staghorn coral critical habitat. Depending on the location of the animals and coral critical habitat in relation to the explosion, there could also be physical impacts to the animals, designated elkhorn and staghorn coral critical habitat, and habitat used by sea turtles and Nassau grouper.

Non-intentional detonations and BIPS may lead to permanent or temporary loss of hearing sensitivity in sperm whales, and leatherback, green, and hawksbill sea turtles; and temporary loss of hearing sensitivity in Nassau grouper. Noise-induced loss of hearing sensitivity or threshold shift refers to an ear’s reduced sensitivity to sound within frequency bandwidths following exposure to different sound sources; when an ear’s sensitivity to sound has been reduced, sounds must be louder for an animal to detect and recognize it. Noise-induced loss of hearing sensitivity is usually represented by the increase in intensity (in decibels) sounds must have to be detected. These losses in hearing sensitivity rarely affect the entire frequency range an ear might be capable of detecting; instead, they affect the frequency ranges that are roughly equivalent to or slightly higher than the frequency range of the noise itself (NMFS 2018a).

For marine mammals in particular, when permanent loss of hearing sensitivity, or PTS, occurs, there is physical damage to the sound receptors (hair cells) in the ear that can result in total or partial deafness, or an animal’s hearing can be permanently impaired in specific frequency ranges, which can cause the animal to be less sensitive to sounds in that frequency range. Traditionally, investigations of temporary loss of hearing sensitivity, or TTS, have focused on sound receptors (hair cell damage) and have concluded that this form of threshold shift is temporary. Hair cell damage does not accompany TTS in these studies and losses in hearing sensitivity were determined to be short-term and are generally followed by a period of recovery to pre-exposure hearing sensitivity that can last for minutes, days, or weeks. More recently, however, Kujawa and Liberman (2009) reported on noise-induced degeneration of the cochlear nerve that is a delayed result of acoustic exposures that produce TTS, that occurs in the absence of hair cell damage, and that is irreversible. They concluded that the reversibility of noise-induced threshold shifts, or TTS, could disguise progressive neuropathology that would have long-term consequences on an animal’s ability to process acoustic information. If this phenomenon occurs in a wide range of species, TTS may have more permanent effects on an animal's hearing sensitivity than earlier studies would lead us to recognize (NMFS 2018a). In addition, there is no way of knowing the severity or degree of TTS an animal sustains from one or multiple exposures, which can either be minor or compounded over time. Therefore, while TTS is generally considered a less severe impairment compared to PTS, over time TTS may result in PTS.

Several variables affect the amount of loss in hearing sensitivity: the level, duration, spectral content, and temporal pattern of exposure to an acoustic stimulus as well as differences in the sensitivity of individuals and species. All of these factors combine to determine whether an individual organism is likely to experience a loss in hearing sensitivity because of acoustic exposure (Ward et al. 1998; Yost 2007). In most circumstances, free-ranging animals are not likely to remain in a sound field that contains potentially harmful levels of noise unless they have a compelling reason to do so (for example, if they must feed or reproduce in a specific location). Any behavioral responses that would take an animal out of a sound field or reduce the intensity of its exposure to the sound field would also reduce the animal's probability of experiencing noise-induced losses in hearing sensitivity (NMFS 2018a). Based on the evidence available from empirical studies of animal responses to human disturbance, marine animals are likely to exhibit one of several behavioral responses upon being exposed to anthropogenic sounds considered in this Opinion: (1) they may engage in horizontal or vertical avoidance behavior to avoid exposure or continued exposure to a sound that is painful, noxious, or that they perceive as threatening; (2) they may engage in evasive behavior to escape exposure or continued exposure to a sound that is painful, noxious, or that they perceive as threatening, which we would assume would be accompanied by acute stress physiology; (3) they may remain continuously vigilant of the source of the acoustic stimulus, which would alter their time budget. That is, during the time they are vigilant, they are not engaged in other behavior; and (4) they may continue their pre-disturbance behavior and cope with the physiological consequences of continued exposure (NMFS 2018a).

Although the published body of science literature contains numerous theoretical studies and discussion papers on hearing impairments that can occur with exposure to a strong sound, only a few studies provide empirical information on noise-induced loss in hearing sensitivity in marine mammals. Hearing loss due to auditory fatigue in marine mammals was studied by numerous investigators (Finneran and Schlundt 2010; Finneran et al. 2010; Finneran et al. 2005; Southall et al. 2007a; Finneran et al. 2000; Finneran et al. 2002; Kastak et al. 2007; Lucke et al. 2009; Mooney et al. 2009b; Mooney et al. 2009a; Nachtigall et al. 2003; Nachtigall et al. 2004; Popov et al. 2011; Schlundt et al. 2000; Southall et al. 2007b; Mann et al. 2010). The studies of marine mammal auditory fatigue were all designed to determine relationships between TTS and exposure parameters such as level, duration, and frequency. In these studies, hearing thresholds were measured in trained marine mammals before and after exposure to intense sounds. The difference between the pre-exposure and post-exposure thresholds indicates the amount of TTS. Species studied include the bottlenose dolphin (nine individuals), beluga (2), harbor porpoise (1), finless porpoise (2), California sea lion (3), harbor seal (1), and northern elephant seal (1). Some of the more important data obtained from these studies are onset-TTS levels—exposure levels sufficient to cause a just-measurable amount of TTS, often defined as 6 dB of TTS (for example Schlundt et al. 2000).

Physical injury is an additional effect to animals, especially for Nassau grouper and sea turtles, that may occur from detonations. The likelihood of internal bodily injury from explosive detonations is related to the received impulse of the underwater blast (pressure integrated over time), not peak pressure or energy (Richmond et al. 1973; Yelverton and Richmond 1981; Yelverton et al. 1973; Yelverton et al. 1975). Therefore, impulse is used as a metric upon which internal organ injury can be predicted. Onset mortality and onset slight lung injury are defined as the impulse level that would result in 1 percent mortality (most survivors have moderate blast injuries and should survive) and zero percent mortality (recoverable, slight blast injuries) in the exposed population, respectively. Criteria for onset mortality and onset slight lung injury were developed using data from explosive impacts on mammals (Yelverton and Richmond 1981; NMFS 2018a).

The impulse required to cause lung damage is related to the volume of the lungs. The lung volume is related to both the size (mass) of the animal and compression of gas-filled spaces at increasing water depth. In terms of gastrointestinal tract (GI) injuries, gas-containing internal organs, such as lungs and intestines, have been shown to be the principle damage sites from shock waves in submerged terrestrial mammals (Ward and Clark 1943; Greaves et al. 1943; Yelverton et al. 1973; Richmond et al. 1973). Slight injury to the GI may be related to the magnitude of the peak shock wave pressure over the hydrostatic pressure and would be independent of the animal’s size and mass (Goertner 1982).

Masking is a phenomenon that affects animals that are trying to receive acoustic information about their environment, including sounds from other members of their species, predators, prey, and sounds that allow them to orient in their environment. Masking these acoustic signals can disturb the behavior of individual animals, groups of animals, or entire populations.

In addition to the potential effects of noise from underwater detonations, depending on the location and magnitude of the explosion in relation to the location of sperm whales, Nassau grouper, and green, leatherback, and hawksbill sea turtles, animals could sustain injury or be killed by propelled fragments. However, studies of underwater bomb blasts show that fragments are larger than those produced during air blasts and decelerate much more rapidly (O'keeffe and Young 1984; Swisdak Jr. and Montaro 1992), reducing the risk to marine organisms. Strikes of animals from munitions fragments resulting from underwater explosions are unlikely based on previous consultations with the Navy for training and testing activities around the U.S.

Similarly, habitat used by green and hawksbill sea turtles and Nassau grouper, as well as elkhorn and staghorn coral critical habitat could suffer damage or destruction due to the physical impacts of a blast and associated blast fragments. Portions of seagrass and coral habitats could be lost should a blast occur in these habitats. Depending on the depth, location, and magnitude of the blast, seagrass roots and rhizomes could be lost or damaged, which would reduce or eliminate the possibility of natural recovery of this habitat, leading to a decrease in seagrass habitat available to green and hawksbill (particularly juvenile) sea turtles and Nassau grouper for foraging. Depending on the location and magnitude of the blast, the hard structure of coral habitat could be lost or damaged. Natural recovery of coral habitat is not expected because the hard structure is the result of the growth and death of organisms with a calcium carbonate structure over many years. This means there could be a decrease in coral habitat available to green and hawksbill sea turtles and Nassau grouper for refuge and foraging and ESA-listed corals for settlement and growth, as well as a loss of the structure and function of a portion of elkhorn and staghorn coral critical habitat within the action area.

Sperm Whales

Marine mammals use sound for communication, feeding, and navigation. To better reflect marine mammal hearing, Southall et al. (2007b) recommended that marine mammals be divided into hearing groups, and NMFS made modifications to these groups to divide pinnipeds into two groups and to re-categorize hourglass and Peale's dolphins (*Lagenorhynchus cruciger* and *Lagenorhynchus australis*, respectively) from mid-frequency to high-frequency cetaceans (NMFS 2016; 2018b; Table 4).

Table 4. Marine Mammal Functional Hearing Groups (Southall et al. 2007b; NMFS 2016)

Hearing Group	Generalized Hearing Range
Low-frequency cetaceans (baleen whales)	7 Hz to 35 kHz
Mid-frequency cetaceans (dolphins, toothed whales, beaked whales, bottlenose whales)	150 Hz to 160 kHz
High-frequency cetaceans (true porpoises, *Kogia*, river dolphins, cephalorhynchid, *Lagenorhynchus cruciger*, *Lagenorhynchus australis*)	275 Hz to 160 kHz
Phocid pinnipeds (underwater) (true seals)	50 Hz to 86 kHz
Otariid pinnipeds (underwater) (sea lions and fur seals)	60 Hz to 39 kHz

The impetus for dividing marine mammals into functional hearing groups was to produce thresholds for each group for the onset of TTS and PTS. The 2016 NMFS guidance and 2018 revisions include a protocol for estimating PTS onset thresholds for impulsive (e.g., airguns, impact hammer pile drivers, explosions) and non-impulsive (tactical sonar, vibratory pile drivers) sound sources. The thresholds serve as a tool to help evaluate the effects of activities employing different sound sources.

The onset of TTS or PTS from exposure to underwater explosions is predicted using sound exposure level-based thresholds in combination with peak pressure thresholds. Based on exposure functions, the onset thresholds for TTS and PTS in sperm whales proposed by the Navy for explosives were developed (NMFS 2018a). The Criteria and Thresholds for Navy Acoustic Effects Analysis Technical Report (U.S. Navy 2017) include non-auditory injury assessments based on exposure thresholds. Increasing animal mass and increasing animal depth both increase the impulse thresholds (i.e., decrease susceptibility; NMFS 2018a). The sound exposure criteria for toothed whales are:

- Onset TTS: 170 dB SEL (weighted) or 224 dB Peak SPL (unweighted)
- Onset PTS: 185 dB SEL (weighted) or 230 dB peak SPL (unweighted)
- Onset injury (impulse):

  Exposure Threshold: $65.8M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$

  Threshold for Farthest Range to Effect: $47.5M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$ Pa-s
- Onset injury (peak pressure):

  Exposure Threshold: 243 dB re 1 μPa SPL peak

  Threshold for Farthest Range to Effect: 237 dB re 1 μPa SPL peak
- Onset mortality (impulse):

  Exposure Threshold: $144M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$

  Threshold for Farthest Range to Effect: $103M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$

Where SEL = sound exposure level; SPL = sound pressure level; M = mass of animals (kg); D = depth of animals (m). The threshold for farthest range to effect is the threshold for one percent risk used to assess mitigation effectiveness.

The echolocation calls of toothed whales are subject to masking by high frequency sound. Studies on captive odontocetes by Au (1993), Au et al. (1985), and Au et al. (1974) indicate that some species may use various processes to reduce masking effects (e.g., adjustments in echolocation call intensity or frequency as a function of background noise conditions). There is also evidence that the directional hearing abilities of odontocetes are useful in reducing masking at the high frequencies these cetaceans use to echolocate, but not at the low-to-moderate frequencies they use for communication (Zaitseva et al. 1980). Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses produced by echosounders and submarine sonar (Watkins and Schevill 1975; Watkins et al. 1985). 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).

As with hearing loss, auditory masking can effectively limit the distance over which a marine mammal can communicate, detect biologically relevant sounds, and echolocate (odontocetes). Unlike auditory fatigue (temporary loss of hearing after exposure to sound resulting in a temporary shift of the auditory threshold or TTS), which always results in a localized stress response, behavioral changes resulting from auditory masking may not be coupled with a stress response. Another important distinction between masking and hearing loss is that masking only occurs in the presence of the sound stimulus, whereas hearing loss can persist after the stimulus is gone (NMFS 2018a).

Given the above, depending on the size of the munitions item that detonates either due to a BIP or unintentionally and the location of the explosion in relation to the location of animals, sperm whales could suffer PTS, TTS, or exhibit behavioral changes such as avoidance, stoppage of echolocation and calling, or fleeing the area.

In terms of the possibility of sperm whales being struck during an explosion, the Navy modeled the potential exposure of sperm whales to fragments from non-explosive practice munitions and high-explosive munitions as part of the ESA section 7 consultation with NMFS for the Atlantic Fleet Training and Testing (NMFS 2018a). The Navy reported that a disturbance or strike as it falls through the water column is not very likely because the objects generally sink through the water slowly and can be avoided by most cetaceans. The Navy also reported that no strike from military expended materials has ever been reported or recorded in the AFTT area, which includes Puerto Rico although no active military training occurs in the U.S. Caribbean, but they used statistical probability modeling to estimate the likelihood. For sperm whales, there was a 0.02 percent probability that a sperm whale could be struck during training exercises and a 0.24 percent probability during testing activities in the Southeast United States Continental Shelf Large Marine Ecosystem and Gulf Stream Open Ocean Area Jacksonville Range Complex (United States Fleet Forces 2009). Thus, there is the potential for sperm whales to be struck by fragments from an underwater explosion but this is more likely for explosions at the water surface in the immediate area where these animals are present when they are present, which is not expected to occur during the activities described in this Opinion. Therefore, the effects of explosions that are likely to result in fitness consequences to a few individuals, likely mother-calf pairs and juveniles, are associated with the noise of the explosion and are discussed further in Section 8.2.4.

<u>Nassau Grouper</u>

All fish have two sensory systems to detect sound in the water: the inner ear, which functions very much like the inner ear in other vertebrates, and the lateral line, which consists of a series of receptors along the fish's body (Popper 2008). The inner ear generally detects relatively higher-frequency sounds, while the lateral line detects water motion at low frequencies (Hastings and Popper 2005).

Studies of the effects of human-generated sound on fish have been reviewed in numerous places (e.g., Hastings and Popper 2005; Popper 2003; Popper 2008; Hastings and Popper 2009; Popper et al. 2004; NRC 1994; Popper and Schilt 2009; Popper and Hastings 2009). Most investigations, however, have been in the gray literature (non-peer-reviewed reports - see Hastings and Popper 2005; Popper 2008; Hastings and Popper 2009 for extensive critical reviews of this material). Studies have been published assessing the effect on fish of short-duration, high-intensity signals such as might be found near high-intensity sonar, pile driving, or seismic air guns, none of which are part of the proposed activities considered in this consultation.

Studies of the effects of long-duration sounds with sound pressure levels below 170 to 180 dB re 1 μPa indicate that there is little to no effect of long-term exposure on species that lack notable anatomical hearing specialization (Amoser and Ladich 2003; Scholik and Yan 2001; Smith et al. 2004b; Smith et al. 2004a; Wysocki et al. 2007). The longest of these studies exposed young rainbow trout (*Onorhynchus mykiss*), to a level of noise equivalent to one that fish would experience in an aquaculture facility (e.g., on the order of 150 dB re 1 μPa) for about 9 months. The investigators found no effect on hearing (i.e., TTS) as compared to fish raised at 110 dB re 1 μPa (Popper et al. 2007).

In contrast, studies on fish with hearing specializations (i.e., greater sensitivity to lower sound pressures and higher frequencies) have shown that there is some hearing loss after several days or weeks of exposure to increased background sounds, although the hearing loss seems to recover (e.g., Scholik and Yan 2002; Smith et al. 2006; Smith et al. 2004b). Smith et al. (2004b); (2006) exposed goldfish to noise at 170 dB re 1 μPa and found a clear relationship between the amount of hearing loss (TTS) and the duration of exposure until maximum hearing loss occurred after 24 hours of exposure. A 10-minute exposure resulted in a 5 dB TTS, whereas a 3-week exposure resulted in a 28 dB TTS that took over 2 weeks to return to pre-exposure baseline levels (Smith et al. 2004; Note: recovery time not measured by investigators for shorter exposure durations).

Concern about potential fish mortality associated with the use of at-sea explosives led military researchers to develop models that predict safe ranges for fish and other animals from explosions of various sizes (see, for instance, Goertner 1982; Goertner et al. 1994; Yelverton et al. 1975). Young (1991) provides equations that allow estimation of the potential effects of underwater explosions on fish possessing swim bladders, which Nassau grouper do, using a damage prediction method developed by Goertner (1982). Young (1991) used the size of the fish and its location relative to the explosive source as parameters but made these independent from environmental conditions such as the depth where the fish is located and explosive shot frequency.

More recently, in consultations with the Navy, NMFS used the mortality criteria provided in the 2014 American National Standards Institute (ANSI) Guidelines (Popper et al. 2014), which divides fish according to the presence of a swim bladder and if the swim bladder is involved in hearing. NMFS also used the Navy's AFTT Phase III BA (Department of the Navy 2017) and the AFTT Final EIS (Department of the Navy 2018) impact pile driving and air gun injury thresholds suggested by the ANSI Guidelines as surrogates for numeric thresholds for injury and

TTS in fish from explosions (NMFS 2018a). This was done because the 2014 ANSI Guidelines did not suggest numeric thresholds for injury or TTS due to explosives for fish. Nassau grouper have a swim bladder but it is not involved in hearing. The species also lacks hearing specializations and primarily detects particle motion at frequencies below 1 kHz (NMFS 2018a). Therefore, the sound exposure criteria for mortality, injury, and TTS from explosives for fish with a swim bladder not involved in hearing (that include Nassau grouper) are:

- Onset TTS: >186 dB SEL_{cum}
- Onset of Injury: 203 dB (SEL_{cum}), >207 dB SPL_{peak}
- Onset of Mortality: 229 dB SPL_{peak}

Where SEL_{cum} = cumulative sound exposure level (dB re 1 μPa^2-s); SPL_{peak} = peak sound pressure level (dB re 1 μPA); and > indicates that the given effect would occur above the reported threshold.

Auditory masking refers to the presence of a noise that interferes with a fish's ability to hear biologically relevant sounds. Fish use sounds to detect both predators and prey, and for schooling, mating, and navigating (Popper 2003). Acoustic stressors during spawning migrations of ESA-listed fish species could lead to behavioral responses or auditory masking that affect an individual's ability to find a mate. Any noise (i.e., unwanted or irrelevant sound, often of an anthropogenic nature) detectable by a fish can prevent the fish from hearing biologically important sounds including those produced by prey or predators (Popper 2003). The frequency of the sound is an important consideration for fish because many marine fish are limited to detection of the particle motion component of low frequency sounds at relatively high sound intensities (Amoser and Ladich 2003).

Of considerable concern is that human-generated sounds could mask the ability of fish to use communication sounds, especially when the fish are communicating over some distance. In effect, the masking sound may limit the distance over which fish can communicate, thereby having an impact on important components of their behavior. For example, the sciaenids, which are primarily inshore species, are one of the most active sound producers among fish, and the sounds produced by males are used to "call" females to breeding sights (Ramcharitar et al. 2001) reviewed in Ramcharitar et al. (2006). If the females are not able to hear the reproductive sounds of the males, there could be a significant impact on the reproductive success of a population of sciaenids. Because most sound production in fish used for communication is generally below 500 Hz (Slabbekoorn et al. 2010), sources with significant low-frequency acoustic energy could affect communication in fish. Nassau grouper produce courtship sounds during spawning aggregations that are species-specific. The calls consist of a pulse train with a varying number of short individual pulses and tonal sound in the 30 to 300 Hz band (Ibrahim et al. 2018). Thus, low-frequency sound sources present during spawning could affect reproductive success.

One of the problems with existing fish auditory masking data is that the bulk of the studies have been done with goldfish, a freshwater fish with well-developed anatomical specializations that enhance hearing abilities. The data on other species are much less extensive. As a result, less is known about masking in marine species, many of which lack the notable anatomical hearing specializations. However, Wysocki and Ladich (2005) suggest that ambient sound regimes may limit acoustic communication and orientation, especially in animals with notable hearing specializations.

Also potentially vulnerable to masking is navigation by larval fish, although the data to support such an idea are still limited. There is indication that larvae of some reef fish (species not identified in study) may have the potential to navigate to juvenile and adult habitat by listening for sounds emitted from a reef (either due to animal sounds or non-biological sources such as surf action; e.g., Higgs 2005). In a study of an Australian reef system, the sound signature emitted from fish choruses was between 0.8 and 1.6 kHz (Cato 1978) and could be detected by hydrophones 3 to 4 nm from the reef (McCauley and Cato 2000). Snapping shrimp in Kaneohe Bay, Hawaii, were found to have clicks with a low-frequency peak between 2 and 5 kHz and energy extending out to 200 kHz (Au and Banks 1998). These bandwidths are within the detectable bandwidth of adults and larvae of the few species of reef fish, such as the damselfish, *Pomacentrus partitus,* and bicolor damselfish, *Eupomacentrus partitus,* that have been studied (Kenyon 1996; Myrberg Jr. 1980). There is also evidence larval fish may be using other kinds of sensory cues, such as chemical signals, instead of, or alongside of, sound (Atema et al. 2002).

Disturbance or strike to Nassau grouper could result from fragments falling through the water column in small areas and fish are expected to leave the area of disturbance prior to any explosions, making the probability of disturbance or strike minimal. The Navy did not model the probability of fragment strike for fish as they did for marine mammals and sea turtles as part of previous consultations for training and testing activities in the U.S., in part because fish are below the water and likelihood of observing an impact is low. In terms of physical damage to habitat, depending on where detonations occur and at what scale, nursery and adult refuge and foraging habitats could be lost or damaged, resulting in fitness consequences to individuals associated with the need to move to different areas containing similar habitats. The effects of explosions that are likely to result in fitness consequences to individual Nassau grouper are discussed further in Section 8.2.4.

Leatherback, Hawksbill, and Green Sea Turtles

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Moein Bartol and Musick 2003; Bartol and Ketten 2006), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Additionally, they are not known to produce sounds underwater for communication.

Available information suggests that the auditory capabilities of sea turtles are centered in the low frequency range (<2 kHz; Bartol et al. 1999; Piniak 2012; Lenhardt et al. 1983; Lenhardt et al.

1994; Ridgway et al. 1969), with greatest sensitivity below 1 kHz. A more recent review on sea turtle hearing and sound exposure indicated that sea turtles detect sounds at less than 1,000 Hz (Popper et al. 2014). Research on leatherback sea turtle hatchlings using auditory evoked potentials showed the turtles respond to tonal signals between 50 and 1,200 Hz in water (maximum sensitivity 100 to 400 Hz; 84 dB re: 1 µPa rms at 300 Hz; Piniak 2012).

For sea turtles, the Navy developed criteria to determine the potential onset of hearing loss, physical injury (non-auditory), and non-injurious behavioral response to detonation exposure using the weighting function and hearing group developed by compiling sea turtle audiograms available in the literature to create a composite audiogram for sea turtles as a hearing group (U.S. Navy 2017). The sound pressure or blast wave produced from a detonation may also induce physical injuries such as external damage to the carapace and internal damage to organs and blood vessels in addition to affecting hearing (NMFS 2018a). The sea turtle impact threshold criteria (NMFS 2018a) are:

- Onset TTS: 189 dB SEL_{cum} (re: 1 µPa^2-s) and 226 dB SPL (re: 1 µPa)(0-peak)
- Onset PTS: 204 dB SEL_{cum} (re: 1 µPa^2-s) and 232 dB SPL (re: 1 µPa)(0-peak)
- Onset injury (impulse):

  Exposure Threshold: $65.8M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$

  Threshold for Farthest Range to Effect: $47.5M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$ Pa-s
- Onset injury (peak pressure):

  Exposure Threshold: 243 dB re 1 µPa SPL peak

  Threshold for Farthest Range to Effect: 237 dB re 1 µPa SPL peak
- Onset mortality (impulse):

  Exposure Threshold: $144M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$

  Threshold for Farthest Range to Effect: $103M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$

Where M = mass of animals (kg); D = depth of animals (m). The threshold for farthest range to effect is the threshold for one percent risk used to assess mitigation effectiveness.

Sea turtles may exhibit short-term behavioral reactions, such as swimming away or diving to avoid the immediate area around a source based on studies examining sea turtle behavioral responses to sound from impulsive sources. Pronounced reactions to acoustic stimuli could lead to a sea turtle expending energy and missing opportunities to forage or breed. In nesting season, near nesting beaches, behavioral disturbances may interfere with nesting beach approach. In most cases, acoustic exposures are intermittent, allowing time to recover from an incurred energetic cost, resulting in no long-term consequence (NMFS 2018a).

The Navy conservatively estimated the possibility of a direct strike to a sea turtle based on the distribution and density estimates they had for species and the number of activities in AFTT that would pose a risk (NMFS 2018a). In order to estimate potential direct strike exposures, the Navy developed a scenario using the sea turtle species with the highest average monthly density in areas where there are the greatest amounts of military expended material. Thus, loggerhead sea turtles in the Virginia Capes and JAX range complexes were used in the Navy model. The Navy's model estimates that in JAX, 0.06 direct strikes during testing activities and 0.03 direct strike exposure during training activities could occur per year (NMFS 2017). Because the Navy did not model the probability of exposure for other sea turtle species, we use the calculations for loggerhead sea turtles to note that there is the potential for sea turtles to be struck by fragments from an underwater explosion. This is more likely for explosions at the water surface in the immediate area if these animals are present in the immediate area, which is not likely to be the case during the activities described in this Opinion.

The habitat effects to green and hawksbill sea turtles from underwater detonations could result to fitness consequences for adults and juveniles associated with the need to move to other areas to find refuge and foraging habitat, depending on the location and magnitude of the detonation. The effects of explosions that are likely to result in fitness consequences to individual green, hawksbill, and leatherback sea turtles are discussed further in Section 8.2.4.

ESA-Listed Corals and Elkhorn and Staghorn Coral Critical Habitat

While there have been some recent studies indicating that coral planulae (larvae) respond to acoustic cues in order to find suitable substrate for settlement, the sound levels, types of sound, and other factors driving settlement habitat selection are not well-understood. Detonations would result in changes in the soundscape for a short period, but the physical disturbance from detonations is likely to be the more significant stressor so we focus our discussion of response on physical disturbance.

Physical disturbance affecting ESA-listed corals and elkhorn and staghorn coral critical habitat could take the form of breakage or abrasion of coral colonies by the blast and/or fragments from munitions items, fracturing of the substrate forming critical habitat, or pulverizing of the substrate by the blast (the extent of which would depend on the size of the item). Depending on the location and force of the blast, ESA-listed corals colonies could be completely lost or experience varying degrees of damage. Damage to ESA-listed coral colonies, depending on the severity, could lead to a reduction in reproduction, as corals would dedicate resources to growth rather than reproduction. In addition, damaged corals are more likely to be susceptible to disease, bleaching, and other stressors, which will increase the potential for mortality and declines in reproduction. Similarly, depending on the location and force of the blast, the structure and function of elkhorn and staghorn coral critical habitat could be lost or significantly altered, which would impact recruitment through a decrease in the availability of habitat for coral settlement and growth in the areas where detonations take place. The effects of explosions that are likely to result in fitness consequences to existing ESA-listed coral colonies, decrease the potential for future recruitment and growth and settlement of recruits, and adversely impact the structure and function of elkhorn and staghorn coral critical habitat are discussed further in Section 8.2.4.

8.2.3.3 Habitat Loss or Damage

The responses of green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals and coral critical habitat to habitat loss or damage associated with underwater detonations were discussed above in Section 8.2.3.2. This section focuses on the responses of ESA-listed corals and elkhorn and staghorn coral critical habitat to habitat loss or damage associated with encapsulation of MEC/MPPEH, installation of in-water structures (such as anchor pins) in coral habitat, and removal actions in areas containing coral substrate where items have become embedded in the substrate.

If there are ESA-listed corals growing within the footprint where encapsulation of an item or items is planned that cannot be transplanted either because of their growth form, the explosive hazard presented by the item, or some other reason, these colonies would be lost. Encapsulation of items in coral habitats that the Navy determines cannot be removed from the substrate but present an explosive hazard would result in the removal of a portion of substrate from use by recruits of ESA-listed corals. Depending on the material used for encapsulation, the area might be recolonized by benthic organisms, including corals, in the future but, at least in the short-term, the encapsulated area would not provide suitable substrate for coral recruitment and growth.

Anchor pins to secure marker buoys or serve as secondary anchors for in-water structures such as floating barriers are the only component of in-water structures currently proposed for installation in hard bottom habitats, including coral reefs. A single anchor pin will have an impact area of 28 in^2. Other in-water structures, such as buoy tackle and floating tackle, may be present in waters in or adjacent to coral habitats containing ESA-listed coral colonies and/or elkhorn and staghorn coral critical habitat. The small footprint where anchor pins are installed would no longer be available for coral recruits to settle and grow. Anchor pins would not be installed in ESA-listed coral colonies or immediately adjacent to these colonies so only future recruits would be affected by the loss of settlement area within the footprint of each anchor pin. ARMS also would not be installed in areas containing ESA-listed coral colonies or designated critical habitat for elkhorn and staghorn corals and so are not expected to affect critical habitat unless they move during storms. These structures will be monitored to ensure their concrete anchors are sufficient to hold them in place in order to avoid any damage to critical habitat or ESA-listed coral colonies.

Similarly, the functionality of elkhorn and staghorn coral critical habitat as settlement habitat would only be eliminated within the small anchor pin footprint if these are located within critical habitat. In-water structures such as floating barriers that are located in such a way as to cause shading of ESA-listed corals could cause the corals to suffer health consequences. A study of the effects of shading by a pier on *Siderastrea siderea* and *Diploria clivosa*, two Caribbean coral species that are considered more tolerant to environmental variability than ESA-listed corals such as elkhorn and staghorn, found tissue growth, calcification, skeletal extension, and mesenterial fecundity were significantly decreased, as well juvenile density for *Siderastrea* *siderea* in the area most affected by shading by the pier (Durant 2006). *Diploria clivosa* in this area also demonstrated a significant decrease in mesenterial fecundity, as well as a significant increase in zooxanthellae density, indicating that the corals may have been attempting to compensate for the decrease in photosynthetic capacity due to lower light availability by increasing the number of photosynthetic organisms in their tissues (Durant 2006). Thus, shading by in-water structures is likely to reduce the growth and reproductive capacity of ESA-listed coral colonies in the shadow of the structures if these structures are relatively fixed rather than in constant motion with the waves and currents (such as buoys).

The removal of items that have become encrusted in hard substrate could result in adverse effects to ESA-listed corals and elkhorn and staghorn coral critical habitat. Depending on the size of the item, the degree to which it has become embedded in hard substrate, and the method of removal, ESA-listed coral colonies could be lost, broken, or abraded if they are within the footprint where the removal activity will take place. However, some ESA-listed coral colonies may be transplanted outside the removal footprint prior to the removal action if feasible. The response of corals to transplant are discussed further in Section 8.2.3.5, so the discussion in this section focuses on the potential loss or damage to ESA-listed corals. Damage to ESA-listed coral colonies, depending on the severity, could lead to a reduction in reproduction, as corals would dedicate resources to growth rather than reproduction. In addition, damaged corals are more likely to be susceptible to disease, bleaching, and other stressors, which will increase the potential for mortality and declines in reproduction. Removal of encrusted items from areas containing the PBF for elkhorn and staghorn coral critical habitat would result in, at a minimum, damage to elkhorn and staghorn coral critical habitat. The function of the area of critical habitat affected by the removal of encrusted items as habitat suitable for settlement and growth of elkhorn and staghorn coral would be lost. Depending on the scale of the removal action, natural recovery of the habitat may not occur.

Overall, habitat loss or damage due to encapsulation, installation of anchor pins and in-water structures will result in a reduction in fitness for affected ESA-listed coral colonies and a reduction in the function of areas containing the PBF for elkhorn and staghorn coral critical habitat. The effects of the fitness consequences to ESA-listed corals and loss of function of areas of critical habitat are discussed further in Section 8.2.4.

8.2.3.4 Bycatch

Biological sampling using cast nets and fish traps will occur in shallower waters within UXO 16 as part of efforts to determine whether or not MEC and MC presence has affected marine organisms through uptake of contaminants or consumption of contaminated food items. Juvenile green and hawksbill sea turtles, some of which are likely residents within the action area, could be caught in fish traps. Juvenile green and hawksbill sea turtles are not likely to be caught in cast nets because these nets are thrown and the person throwing the net would be able to see whether sea turtles were in the area prior to making a throw. However, if sea turtles were caught in the cast net, because the net is thrown and then immediately pulled, sea turtles would be freed rapidly from the net. Small sea turtles are more likely to swim into fish traps, if the opening is large enough. Sea turtles that are caught in fishing gear used during biological sampling activities could suffer injuries such as cuts, scrapes, and bruises but because the gear would be monitored, these injuries are not expected to be severe and no mortalities are expected. Respiratory and metabolic stress from forcible submergence is correlated with factors such as the size and activity of the sea turtle, water temperature, and biological and behavioral differences between species, all of which affect the survivability of an individual turtle. Sea turtles that are forcibly submerged for longer periods are more susceptible to metabolic acidosis because of high blood lactate levels as lactate levels increase during forced submergence. Lactate levels increase faster in smaller turtles and in warmer areas where routine metabolic rates are higher. The Navy proposes checking traps regularly to be sure turtles have not entered, and this SOP was included in the PDCs for this consultation. Therefore, if periods of forced submergence are short, then the recovery period for affected individuals will be short (NMFS 2017). Handling of turtles in order to free them from fishing gear can result in raised levels of stress hormones, but we expect these effects to be short-term and less than the effects of forced submergence. There will be fitness consequences for individual sea turtles that suffer forced submergence due to entanglement or entrapment in fishing gear used during biological sampling as well as handling in order to free them from the gear. These consequences are discussed further in Section 8.2.4.

Juvenile Nassau grouper that use shallow habitats could be caught in cast nets and traps. Cast net sampling will target areas with schools of fish, but early juveniles (approximately 4.5 – 15 cm TL) and juveniles are relatively solitary. The use of cast nets could result in the capture or juvenile Nassau grouper depending on the benthic habitats where these nets are used. Juvenile Nassau grouper could also swim into fish traps. When cast nets are hauled out of the water or fish traps pulled up, juvenile Nassau grouper could be returned to the water. However, not all fish returned to the water will survive depending on the length of time the net or trap is out of the water and whether any of the gear has caused injury to the fish. In addition, if the people doing the sampling do not properly identify juvenile Nassau grouper, individuals of this species captured during biological sampling could be grouped with other fish species retained for further analysis. Thus, there could be mortality of some juvenile Nassau grouper and fitness consequences to other individuals associated with bycatch in cast nets and fish traps. These consequences are discussed further in Section 8.2.4.

The BA discusses other types of nets, the use of which could result in bycatch of juvenile sea turtles and Nassau grouper. As explained in this Opinion, because the BA specifies that only cast nets will be used, the use of other types of nets and their effects on ESA-listed species were not analyzed in the Opinion. Reinitiation of consultation may be required if the Navy decides to use any net type other than cast nets and the effects of the use of these nets on ESA-listed species would differ from the effects analyzed for the use of cast nets.

8.2.3.5 Entrapment

Hatchling leatherback, green, and hawksbill sea turtles could congregate along in-water structures such as floating barriers and may be less likely than larger sea turtles to dive under the barriers. Hawksbill hatchlings tend to swim slowly away from the beach and shelter in floating algal mats and other marine detritus (Chung et al. 2009). Green and leatherback hatchlings tend to swim almost continuously for the first 24-hours once they reach the water after emerging from their nests (Wyneken and Salmon 1992). Green sea turtle hatchlings shelter in floating algal mats and other marine detritus. Depending on wave and current patterns, marine detritus could concentrate along in-water structures, which could make them more attractive to hatchlings. Wave and current patterns could also push hatchlings to the structures. In-water barriers installed at Bahia Icacos (but no longer in the water) included a barrier maintenance and monitoring to remove accumulated debris in order to maintain the integrity of the structures, as well as to relocate sea turtle hatchlings found along the barriers during nesting season. However, there was no nesting recorded on the beach in Bahia Icacos during the period the barriers were in place so no sea turtle hatchlings were observed along the barriers.

If in-water structures have lights, they can cause disorientation of hatchlings, leading to potentially greater numbers of animals moving toward and congregating at the structures. Stewart and Wyneken (2004) reported in-water hatchling survival rates of 95 percent along a natural nesting beach in Florida. Other studies have shown that hatchling mortality rates range from 30 to 60 percent as the animals leave the beach and swim toward open water, and only 2.5 in 1,000 reach adulthood (Pilcher 1999; Frazer 1992). Hatchling sea turtles are preyed upon by large predatory fishes such as jacks, tarpon, barracuda, and grouper, all of which are present in the action area, as they attempt to reach the open ocean (Stewart and Wyneken 2004; Whelan and Wyneken 2007). Congregation of sea turtle hatchlings along in-water structures would make them more vulnerable to predation by seabirds and other marine organisms. Studies have shown that predation rates are much higher when hatchlings are concentrated in a particular area (Wyneken 2000; Wyneken and Fisher 1998) and marine predators are known to learn to wait at locations where hatchlings concentrate (Wyneken 2000). The average predation rate reported by Stewart and Wyneken (2004) for loggerhead sea turtle hatchlings off the coast of Southeast Florida was five percent. Thus, there will be an increased risk of predation and fitness consequences to individual hatchling leatherback, green, and hawksbill sea turtles due to entrapment by in-water structures. These consequences are discussed further in Section 8.2.4.

8.2.3.6 Organism Collection and Transplant

Up to 50 ESA-listed coral colonies may be sampled over the 20-year lifetime of the action. The Navy indicates that these corals may be cored or sampled in a different manner to collect tissue samples. Sampling will not result in the loss of entire colonies, only portions of tissue. However, tissue removal may make sampled corals more susceptible to disease while the coral is still regrowing. Sampled corals would also be more susceptible to bleaching if sampling events take place prior to periods of elevated sea surface temperatures. If corals become diseased or bleach, they could suffer full or partial mortality. If sampled corals are sexually mature, bleaching or disease is likely to result in these corals not reproducing at a minimum in the year when the stress occurs. While the Navy was unable to provide information regarding the number of colonies of each species that will be sampled, percentages of each species found in surveys conducted around Vieques indicate that mountainous and boulder star corals and staghorn coral are the most common species in the areas where surveys were conducted. Dominance by star coral species is common in Puerto Rico and the Virgin Islands, so it is likely that these are the most common hard coral species around Vieques as well. If we use the percentages of corals found during surveys, it is likely that the 50 ESA-listed corals sampled during the 20-year timeframe of the action would be boulder star, mountainous star, and/or staghorn coral colonies.

ESA-listed corals could be affected by removal activities including collection and transplant of ESA-listed coral colonies growing on items to be removed or within the footprint of areas to be encapsulated. Removal of ESA-listed coral colonies from some proposed in-water structure footprints will require step-down consultation. Most in-water structures constructed by the Navy to date have been floating structures and the anchor systems for most structures are located outside coral habitats. The Coral Ark structures may be used for transplant of ESA-listed corals to the structures, which will be evaluated in future step-down consultations once details of these structures are known. All removal and transplant activities are expected to have the same effects on ESA-listed coral colonies. Not all ESA-listed coral colonies will be candidates for removal and transplant and those that are may be only partially removed from an item or impact footprint for transplant. Whether a coral colony can be removed completely for transplant will depend on the size of the colony and its growth form (i.e. encrusting versus other forms), as well as the stability of the area it has colonized and safety risks associated with disturbance of MEC/MPPEH. Any portions of a colony left behind are expected to suffer mortality. We expect there could be 10 percent mortality of transplanted corals based on coral transplant work in Puerto Rico, such as that for the USACE San Geronimo restoration project in the Condado Lagoon, San Juan, Puerto Rico in 2006. Transplanted corals could also suffer temporary declines in health due to the stress of transplantation. Temporary declines in the health of coral colonies that survive transplantation would be evidenced by bleaching and/or partial tissue mortality, and a lack of sexual reproduction within the first spawning season following transplantation.

8.2.4 **Risk Analysis**

As discussed in previous sections, we believe several of the activities that are part of the action, as well as non-intentional detonation that may occur as a result of some of these activities, is likely to result in potential injury to sperm whales, green, hawksbill, and leatherback sea turtles, Nassau grouper, and ESA-listed corals; potential behavioral responses in sperm whales, sea turtles, and Nassau grouper; and potential loss or degradation of elkhorn and staghorn coral critical habitat. The consequences of these responses are discussed further below.

8.2.4.1 Equipment Collisions

The annual potential mortality or decrease in fitness of ESA-listed coral colonies due to equipment collisions likely includes the same colonies within UXO 16 that could be impacted by removal activities, including those involving relocation of ESA-listed corals. The only time a collision with equipment was reported during an underwater survey, two ESA-listed coral colonies were fragmented by the collision. In terms of the potential impact of fitness consequences to a few ESA-listed coral colonies in years when underwater equipment is used, as stated previously, mature colonies might not spawn the year in which breakage occurs due to the stress of severe breakage. Similarly, colonies affected by breakage that bleach or become infected by a disease would not spawn that year and could be lost from the population if bleaching or disease is severe enough to cause full or partial mortality of the colonies. Any colonies that suffer mortality because of collisions or due to stressors that are more likely to affect impacted corals would be removed from the pool of reproductive individuals in the action area. However, we believe the fitness consequences to or loss of up to two (if we assume the same level of effects as during the one year when an equipment collision with ESA-listed coral colonies did occur) ESA-listed coral colonies annually from collisions with equipment will not have a measurable effect on the population because there are estimated to be thousands of colonies in the action area based on surveys by the Navy and NOAA, including outside areas where underwater MEC/MPPEH has been documented in UXO 16. Thus, the proposed action is not likely to reduce the population viability of ESA-listed corals in the action area.

8.2.4.2 Underwater Detonations

The behavioral effects to sperm whales, green (North and South Atlantic DPS), hawksbill, and leatherback sea turtles, and Nassau grouper as a result of strong responses to underwater detonations are likely to reduce the fitness of a proportion of individual animals that react strongly to the noise from detonation. For these individuals, strong behavioral responses will have energetic consequences that could reduce (at least temporarily) an animal’s growth and hinder reproduction in sexually mature animals or slow growth in immature animals, as well as have health consequences such as weight loss and greater susceptibility to disease and predation. For sperm whale mother-calf pairs, significant behavioral responses on the part of the mother could affect milk production and feeding, impacting the growth and health of the calf. Depending on the severity of TTS, there could be long-term consequences to sperm whales, sea turtles, and Nassau grouper because of underwater detonations. If TTS is severe enough, it could eventually result in PTS for marine mammals and sea turtles or, for animals that have already suffered multiple TTS, these animals could suffer PTS from lower sound levels. Similarly, if the detonation results in PTS, the ability of individual animals to feed, avoid predators, avoid vessels, and communicate, depending on the species affected, would lead to long-term reductions in fitness of individuals.

The last NMFS stock assessment that discussed sperm whales was in 2010, but the U.S. Caribbean stock numbers were identified as unknown. Therefore, it is not possible for us to use the stock assessment report to evaluate the effects of decreases in fitness because of underwater detonations over the expected 20-year consultation period on the population of sperm whales in the U.S. Caribbean, at least seasonally. Instead, we will assume a worst-case scenario of at least one mother-calf pair and one juvenile sperm whale in the action area during underwater detonations that occur during the winter migration period of these animals from approximately November to March compared to the best population estimate of 763 for the Gulf of Mexico stock (Hayes et al. 2019). If we assume sperm whales will suffer fitness consequences in years when underwater detonations occur during months when this species is present in the action area, and that the stock of sperm whales that includes the U.S. Caribbean is at least the size of that in the Gulf of Mexico, we conclude the fitness effects annually and cumulatively over the 20-year consultation period will not have a measurable effect on the population and are not likely to reduce the population viability of sperm whales. The actual numbers and life stages of sperm whales will be calculated as part of step-down consultations for removal activities that will use BIPs or that propose the recovery of items that are determined to be unstable and thus present a detonation risk.

In terms of the potential impact of fitness consequences to green, hawksbill, and leatherback sea turtles because of significant disturbance, mortality, or injury associated with underwater detonations, we consider the population effects in the context of total annual mortality associated with human activities and the estimated populations of these species. Adult and hatchling leatherback, and adult, juvenile, and hatchling green and hawksbill sea turtles could suffer mortality or fitness consequences because of underwater detonations. The North Atlantic DPS of green sea turtles is estimated to have 167,424 nesting females and the South Atlantic DPS to have 63,332. It is estimated that 22,004 to 29,035 female hawksbill sea turtles nest globally. The population of leatherback sea turtles in the North Atlantic is estimated to be 34,000 to 94,000 adults. Based on nesting data for Vieques, we estimated that there could be 106 adult, up to 4,788 hatchling, and 1,751 juvenile green sea turtles; 30 adult and up to 2,295 hatchling leatherback sea turtles; and 48 adult, up to 3,898 hatchling, and up to 4,377 juvenile hawksbill sea turtles in the action area. The number and life stage of sea turtles of each species that suffer fitness consequences because of underwater detonation will depend on the location where underwater BIPs are planned or where non-intentional detonation may occur during cleanup activities due to the instability of underwater munitions. Sea turtle hatchlings of each species will be affected only if detonations occur at a time of year when hatchlings are emerging from their nests and entering the sea, meaning they would be in waters of UXO 16 during removal activities. Similarly, adult leatherback sea turtles will be affected only if detonation occurs during mating and nesting season, which peaks from May to July around Puerto Rico. We anticipate that a small percentage of the individuals of each life stage of the three sea turtle species that may be present during underwater detonations would be affected. Detonations would occur in localized areas and, based on underwater surveys conducted to date, there are limited numbers and locations where large MPPEH items are present that would result in a larger potential area of influence for acoustic impacts. Therefore, we conclude the fitness effects to different life stages of leatherback, green, and hawksbill sea turtles in years when underwater detonations occur as a result of the proposed action will not have a measurable effect on the population and are not likely to reduce the population viability of the North and South Atlantic DPSs of green sea turtles, leatherback sea turtles, and hawksbill sea turtles. The actual numbers and life stages of individuals of each sea turtle species likely to suffer fitness consequences will be calculated as part of step-down consultations for removal activities that will use BIPs or that propose the recovery of items that are determined to be unstable and thus present a detonation risk.

In order to assess the potential impacts of fitness consequences to Nassau grouper because of significant disturbance, mortality, or injury associated with underwater detonations, we consider the population effects in the context of total annual mortality associated with human activities and the estimated populations of this species. Nassau grouper was once naturally abundant in areas with large shelf habitat, including in the Greater Antilles (which includes Puerto Rico) and evidence indicates there is strong genetic differentiation among subpopulations in the Caribbean (Jackson et al. 2014a). Based on the decline in spawning aggregations estimated as 60 percent over the period from 1980 to 2016, the population was estimated at 3,000 individuals in 2016 (Sadovy et al. 2018) and was expected to continue declining in some areas due to continued fishing pressure. Fisheries data from Puerto Rico based on commercial landings indicate a 99 percent decline in landings from 1998 to 2011, meaning the remaining population of Nassau grouper around Puerto Rico may be very small, though limited population growth is expected now that there is a ban on fishing this species in Commonwealth and Federal waters. The number and life stage of Nassau grouper that suffer fitness consequences because of underwater detonation will depend on the location where underwater BIPs are planned or where non-intentional detonation may occur during cleanup activities due to the instability of underwater munitions. We anticipate that a small percentage of the individuals of juvenile and adult life stages that may be present during underwater detonations would be affected because detonations would occur in localized areas and, based on underwater surveys conducted to date, there are limited numbers and locations where large MEC/MPPEH items are present that would result in a larger potential area of influence for acoustic impacts. Therefore, we conclude the fitness effects to Nassau grouper in years when underwater detonations occur because of the proposed action will not have a measurable effect on the population and are not likely to reduce the population viability of the species. The actual numbers and life stages of individuals likely to suffer fitness consequences will be calculated as part of step-down consultations for removal activities that will use BIPs or that propose the recovery of items that are determined to be unstable and thus present a detonation risk.

The Navy estimated that 5,198 ESA-listed coral colonies within UXO 16 could be affected by removal actions. A portion of these colonies would be affected by any underwater detonations, either as part of BIPs or due to nonintentional detonations. The potential impacts of fitness consequences to ESA-listed corals as a result of mortality or damage associated with underwater detonations are assessed in the context of the estimated populations of each ESA-listed coral species, but it is important to note that we do not have sufficient data to know how many of each species are present in any given location within UXO 16 and the larger action area. The population of elkhorn coral is estimated as hundreds of thousands, of staghorn coral as tens of millions, and of lobed star, boulder star, and mountainous star corals as millions of colonies of each species. We anticipate that a small percentage of the total number of ESA-listed coral colonies in locations where underwater detonations occur would be lost or damaged, and their reproductive potential and associated new recruits would be lost either temporarily, as colonies recover from the stress of damage, or permanently in the case of colonies that are destroyed by the blast. Some species, such as pillar coral and rough cactus corals, are naturally rare while others, such as the three star coral species, are more common. Some species are more common in shallow waters, such as elkhorn coral and pillar coral, while others may be present in deeper waters further offshore. Thus, as for other species discussed in this section, the number, species, and life stage (recruit or sexually mature adult) of ESA-listed corals affected by underwater detonations will depend on the location and magnitude of the blast. As noted previously, the Navy estimates that approximately 5,173 ESA-listed coral colonies are present on or immediately adjacent to MEC/MPPEH. These include the coral colonies that would be affected by underwater detonations. Given that no underwater detonations have occurred during underwater cleanup activities conducted to date and that BIPs are unlikely to be used, in addition to the likelihood that many of the ESA-listed coral colonies on and adjacent to MEC/MPPEH will be transplanted prior to movement of items, be believe underwater detonations leading to mortality of a small subset of the ESA-listed corals in the action area will not have a measurable effect on the population of ESA-listed corals in the action area and is not likely to reduce the population viability of ESA-listed corals in the action area. The actual numbers of coral colonies likely to suffer fitness consequences will be calculated based on the estimated area of impact from detonations as part of step-down consultations for removal activities that will use BIPs or that propose the recovery of items that are determined to be unstable and thus present a detonation risk. .

The WAA found that almost half the benthic habitats within UXO 16 are coral habitats. A determination was not made of how much of this area contains the PBF for elkhorn and staghorn coral critical habitat. Given the depths within UXO 16 and the information from the Navy indicating that approximately half of the coral habitats mapped during the WAA contain coral reef formations, it is likely that elkhorn and staghorn coral critical habitat is present in a significant portion of the action area. The PBF for designated critical habitat for elkhorn and staghorn corals will no longer be present/functional within the footprint of detonations that occur in areas containing coral critical habitat. Natural recovery of the areas within the detonation footprint is not expected. The actual area of elkhorn and staghorn coral critical habitat likely to be lost due to underwater detonations will be calculated as part of step-down consultations for removal activities that will use BIPs or that propose the recovery of items that pose an explosive hazard. However, the detonation footprint and frequency of underwater detonations are expected to be extremely small during the 20-year period of proposed cleanup activities.

8.2.4.3 Habitat Loss or Damage

The annual potential mortality or decrease in fitness of ESA-listed coral colonies due to habitat loss or damage to the corals themselves from encapsulation, in-water structures, and removal of encrusted items likely includes the same colonies within UXO 16 that could be impacted by other stressors associated with the action that are discussed in the other subsections of Section 8.2. To date, encapsulation has not been considered as an option for treatment of munitions that pose a human safety risk but, because it may be used in the future, we have included it in the Opinion. Only the ESA-listed coral colonies within the encapsulation footprint or the footprint of removal activities for encrusted items that cannot be transplanted would be lost from the population. Similarly, only those colonies within the area shaded by in-water structures would suffer fitness consequences. To date, the location of in-water structures such as floating barriers in Bahia Icacos was such that shading did not affect ESA-listed coral colonies. In terms of the potential impact of fitness consequences to a limited number ESA-listed coral colonies when encapsulation and/or removal of encrusted items are used in coral habitats where these colonies are present, or in areas affected by shading from in-water structures, as stated previously, mature colonies might not spawn the year in which breakage or damage occurs due to stress. Similarly, colonies affected by breakage or damage that bleach or become infected by a disease would not spawn that year and could be lost from the population if bleaching or disease is severe enough to cause full or partial mortality. Any colonies that suffer mortality would be removed from the pool of reproductive individuals in the action area. In addition, the loss of settlement habitat within the footprint of encapsulation, removal activities where items are encrusted, or in-water structures such as anchor pins could reduce the number of future recruits in areas affected by the action. However, we believe the fitness consequences to a small number of ESA-listed coral colonies annually from encapsulation, in-water structures, and/or removal of encrusted items will not have a measurable effect on the population and is not likely to reduce the population viability of ESA-listed corals in the action area.

In terms of elkhorn and staghorn coral critical habitat, the PBF for designated critical habitat for elkhorn and staghorn corals will no longer be present/functional within the footprint of encapsulation, anchor pins, or areas where removal of encrusted items occurs. Natural recovery of the areas within the footprint of encapsulation and removal of encrusted items is not expected, although ESA-listed corals may begin to colonize encapsulated areas over time. The area of elkhorn and staghorn coral critical habitat impacted by the installation of an anchor pin is 28 in^2, meaning a large number of anchor pins would have to be installed in the same location in order to result in a measurable loss of habitat area. The actual area of elkhorn and staghorn coral critical habitat likely to be lost due to encapsulation and/or removal of encrusted items will be calculated as part of step-down consultations for removal activities. However, the Navy does not expect to use these removal methods frequently during the 20-year period of proposed cleanup activities.

8.2.4.4 Bycatch

Bycatch of juvenile sea turtles is not expected to result in mortality of any individual green or hawksbill sea turtles. There will be fitness consequences for individuals captured in fishing gear in the form of short-term stress responses and temporary changes in metabolism but we do not expect long-term effects to individuals. The use of cast nets and fish traps for biological sampling will be confined to small areas within UXO 16. In addition, before a cast net is thrown, the user of the net will be able to see whether turtles are present and avoid throwing the net until animals leave the area. For fish traps, there will be checks of the gear every 15 minutes to ensure no sea turtles have been trapped. If we assume the sampling area will be similar to past studies to characterize the fish community around Vieques (Bauer and Kendall 2010), then an area measuring approximately 7,500 m^2 (1.85 ac; 100 m^2 at 75 different sites around Vieques) would be used for biological sampling. No biological sampling with cast nets and fish traps will take place in coral habitats according to the information provided by the Navy thus the 1.85 ac area where biological sampling will occur will be in the approximately 5,422 ac within UXO 16 that do not contain coral habitats (based on the WAA habitat characterization). Wershoven and Wershoven (1992) estimated there were 5 green sea turtles per acre and Diez and Van Dam (2002) estimated up to 0.5 hawksbill sea turtles per acre in their studies of juvenile foraging habitat and home range. Thus, approximately 10 juvenile green sea turtles and 1 juvenile hawksbill sea turtle could suffer fitness consequences as a result of bycatch in cast nets and fish traps used for biological sampling. As discussed in Section 8.2.3, we do not expect animals to suffer mortality. We estimate there could be 1,751 juvenile green sea turtles and up to 4,377 juvenile hawksbill sea turtles in UXO 16 where biological sampling will occur. Therefore, we conclude the fitness effects to juvenile green and hawksbill sea turtles in years when biological sampling occurs will not have a measurable effect on the population and are not likely to reduce the population viability of the North and South Atlantic DPSs of green sea turtles and hawksbill sea turtles.

Juvenile Nassau grouper may be captured in cast nets and fish traps used for biological sampling. Fish caught in cast nets in particular could suffer injury prior to be being released or, as discussed in Section 8.2.3, misidentification of Nassau grouper could lead to mortality of individuals that are grouped with other fish species for sampling. Previous fish surveys (see Bauer et al. 2008; Bauer and Kendall 2010) largely reported only observations of this species rather than numbers, but Bauer and Kendall (2010) did report observing two individuals of this species outside one of the 75 sampling stations. We are assuming the area of biological sampling will be the same as that used for previous fish surveys, or 1.85 ac, meaning 2 Nassau grouper could be affected by fishing gear in years when biological sampling takes place. The population of Nassau grouper was estimated at 3,000 individuals in 2016 (Sadovy et al. 2018) and was expected to continue declining in some areas due to continued fishing pressure. Despite the potential mortality of the two individual juvenile Nassau grouper in years when biological sampling occurs (if the species is misidentified) or fitness consequences due to capture in cast nets or fish traps, we conclude the fitness effects to juvenile Nassau trouper will not have a measurable effect on the population and are not likely to reduce the population viability of Nassau grouper.

8.2.4.5 Entrapment

Based on historic nesting data (Matos et al. 1992; Belardo and Matos 1993; Belardo et al. 1994; Belardo et al. 1995; Belardo et al. 1996; Belardo et al. 1997; Belardo et al. 1998;1999;2000;2001), most beaches have only a few nests but there are certain beaches that appear to be preferred by different sea turtle species that have larger numbers of nests (see Figure 14 for the distribution of nesting beaches used by different species). If we use three as the average number of nests of each species on a particular beach (and assume that in-water structures that may cause entrapment of sea turtle hatchlings will not be installed in waters off the most-used nesting beaches), we can calculate the approximate number of hatchlings affected by the action. Using the method described in Section 7.1.2, we calculate there could be 171 green sea turtle hatchlings of which between 51.3 and 103 (30 to 60 percent; Pilcher 1999; Frazer 1992) may reach the water after emerging from the nest; between 90 and 166 leatherback hatchlings of which between 27 and 100 may reach the water; and 336 hawksbill hatchlings or which between 101 and 202 may reach the water. Thus, up to 103 green hatchlings, 100 leatherback hatchlings, and 202 hawksbill hatchlings could reach the in-water structure. If hatchlings congregate landward along an in-water structure, they will be transported seaward away from the structure. We do not anticipate that hatchling transport will result in mortality of any animals based on hatchling relocation information from nesting programs in the Virgin Islands (Sandy Point and Buck Island, St. Croix) and Florida. Hatchlings that are moved away from the structure are expected to continue moving out to sea once they are relocated. However, because the congregation of sea turtles along an in-water structure and monitoring and relocation activities will not necessarily occur simultaneously, we use the average predation rate reported by Stewart and Wyneken (2004) for loggerhead sea turtle hatchlings off the coast of Southeast Florida of five percent to calculate the number of hatchlings that could be lost to predation while congregating. Using this percentage, 5 green and leatherback hatchlings and 10 hawksbill sea turtle hatchlings could be predated while at the in-water structure. The congregation of hatchlings and associated predation and relocation would occur over whatever number of years the in-water structure was present seaward of sea turtle nesting beaches, if nesting occurs on the beach annually, which is not always the case. Based on more recent sea turtle nesting data the Navy obtained from the USFWS, we estimate there are up to 4,788 green, 2,295 leatherback, and 3,898 hawksbill sea turtle hatchlings in the action area annually. Therefore, we conclude the fitness effects to hatchling green, leatherback, and hawksbill sea turtles and mortality due to predation associated with congregation along in-water structures will not have a measurable effect on the populations of these species and are not likely to reduce the population viability of the North and South Atlantic DPSs of green sea turtles, leatherback, and hawksbill sea turtles.

8.2.4.6 Organism Collection and Transplant

The decrease in fitness of 50 ESA-listed coral colonies, likely staghorn, mountainous and/or boulder star corals, associated with tissue sample collection could result in mature colonies not spawning for at least one spawning cycle due to temporary declines in health from tissue collection. Corals could also suffer full or partial mortality if increased susceptibility to disease and bleaching due to the stress results in full or partial death of the colony. The sample collection methodology would influence the degree to which corals may be rendered more susceptible to stressors such as disease or high sea surface temperatures that can cause bleaching. However, we believe the fitness consequences to 50 ESA-listed coral colonies over the 20-year lifetime of the action, which are the same corals within the action area that will be affected by other activities that are part of the action, will not have a measurable effect on the population and is not likely to reduce the population viability of ESA-listed corals in the action area because there are estimated to be thousands of colonies in the action area based on surveys by the Navy and NOAA, including outside areas where underwater MEC/MPPEH has been documented in UXO 16.

As stated previously, the Navy estimated there are 5,173 ESA-listed coral colonies in UXO 16 that may be affected by the action. Most of these colonies are likely to be star coral species and acroporids and most are likely to be affected by removal activities and, in many cases, collection and transplant to remove ESA-listed corals from the impact footprint prior to MEC/MPPEH removal activities. Only 10 percent of transplanted colonies are expected to die due to the stress of transplant while the rest will suffer temporary effects, including to reproduction. Thus, the fitness consequences to individuals include the temporary loss of reproductive potential for corals that survive transplant. A determination of the approximate number and species of ESA-listed corals that will be transplanted will be provided as part of step-down consultations for particular removal actions. However, we believe the fitness consequences to approximately 5,173 ESA-listed coral colonies, including the potential mortality of 10 percent of these, or 517 corals (if we assume all of the corals the Navy estimates are likely to be adversely affected by the proposed action largely due to their growth on MEC/MPPEH), will not have a measurable effect on the population of ESA-listed corals in the action area and is not likely to reduce the population viability of ESA-listed corals in the action area.

8.2.5 **Programmatic analysis**

In the previous sections we evaluated the exposure, response, and risk to ESA-listed sperm whales, leatherback, green (North and South Atlantic DPSs), and hawksbill sea turtles, Nassau grouper, and corals, and elkhorn and staghorn coral designated critical habitat as a result of the proposed action. In this section we evaluate whether the implementation of the applicable PDCs is sufficient to ensure that the action will not increase the risk to ESA-listed species or designated critical habitat associated with the implementation of the proposed action over the 20-year consultation lifetime.

Most of the PDCs in this Opinion were developed by the Navy in coordination with NMFS based on SOPs used during past surveys and removal actions that did not involve any take of ESA-listed species or damage to designated critical habitat. Additional PDCs were added to this Opinion for activities that have not been conducted by the Navy in the action area to date and are based on past consultations NMFS has conducted for similar activities. It is important to consider that, while the consultation covers a 20-year period, most of the activities conducted over this period are those that produce stressors that we do not expect to result in adverse effects to ESA-listed species or designated critical habitat. With the implementation of the PDCs, other activities that could result in adverse effects will avoid or minimize potential effects to ESA-listed species and designated critical habitat for elkhorn and staghorn corals to a level that is not likely to result in adverse effects. Of the activities that will produce stressors that may result in adverse effects, specifically underwater detonations, coral habitat loss and damage, bycatch, entrapment, and organism collection and transplant, only transplantation of organisms, including ESA-listed corals, is expected to occur frequently. The transplant of corals from underwater munitions to coral habitat is expected to ultimately benefit ESA-listed corals because it will minimize the loss of colonies from the populations within the action area. NMFS regularly recommends that projects whose footprints contain ESA-listed corals include a transplant plan to relocate corals prior to any construction. For activities that produce stressors that may result in adverse effects to ESA-listed sperm whales, sea turtles, and corals, the implementation of the PDCs will reduce the effects of the proposed action such that we do not expect any effects to have population-level consequences over the 20-year lifetime of the proposed action. This reduction of impacts to ESA-listed species due to the implementation of the PDCs further supports our conclusions in Section 8.2.4 that stressors resulting in adverse effects to ESA-listed sperm whales, sea turtles, and corals will not result in measurable effects to the populations of these species in the action area or reduce their population viability in the action area. Similarly, the implementation of the PDCs will reduce the effects of the action on the PBF for elkhorn and staghorn coral critical habitat in order to maintain the function of the habitat and, thus, its conservation value.

8.2.6 Summary of the Effects of the Action on Sperm Whales, Green (North Atlantic and South Atlantic DPSs) Sea Turtles, Leatherback Sea Turtles, Hawksbill Sea Turtles, Nassau Grouper, Elkhorn Coral, Staghorn Coral, Pillar Coral, Rough Cactus Coral, Lobed Star Coral, Boulder Star Coral, Mountainous Star Coral, and Elkhorn and Staghorn Coral Critical Habitat

The implementation of the action, particularly surveys and removal actions that include the use of underwater equipment, the potential use of BIPs, nonintentional detonations, encapsulation, removal of encrusted items, use of cast nets and fish traps, in-water structures, and collection and transport of organisms, is expected to result in the take of sperm whales, green, leatherback, and hawksbill sea turtles, Nassau grouper, and ESA-listed corals, and effects to elkhorn and staghorn coral critical habitat.

The Navy estimates that 5,173 ESA-listed corals will be affected by the action because of their proximity to or growth on MEC/MPPEH. Data are not available that would enable us to accurately determine the number of colonies of each species of listed coral included in this estimate though boulder star, mountainous star, and staghorn corals appear to be the most abundance in UXO 16 based on data from previous surveys. We anticipate the same colonies would be affected by each of the activities that are expected to result in take of ESA-listed corals.

As discussed in the previous sections, we estimate that equipment collisions during underwater surveys and removal activities could result in the take of 2 ESA-listed coral colonies annually, most likely boulder, mountainous star, and/or staghorn coral colonies based on the abundance of these corals in the action area.

We believe the use of cast nets and fish traps could result in the bycatch of 10 juvenile green sea turtles and 1 juvenile hawksbill sea turtle in years when this gear is used for biological sampling. The use of this gear could also result in bycatch of 2 juvenile Nassau grouper in years when biological sampling occurs.

In-water structures such as floating barriers could result in entrapment of 103 green sea turtle hatchlings, 100 leatherback hatchlings, and 202 hawksbill hatchlings annually, assuming structures are placed off beaches with an average of three nests of each species. Entrapped hatchlings are likely to suffer predation, resulting in the mortality of 5 green and 5 leatherback hatchlings and 10 hawksbill sea turtle hatchlings.

Biological sampling of 50 ESA-listed coral colonies, most likely boulder star, mountainous star, or staghorn corals based on the abundance of these in the action area, over the 20-year project timeframe will occur. Organism collection and transport involving the relocation of ESA-listed coral colonies or fragments of colonies from areas where removal activities or in-water structure construction will occur and could affect the 5,173 ESA-listed coral colonies estimated by the Navy to be growing on or in the immediate vicinity of MEC/MPPEH. If all of these corals were transplanted, 517 would be expected to suffer mortality due to transplant stress.

Underwater detonations due to BIPs or unintentional detonation of MEC/MPPEH during removal activities could also result in take of sperm whales, likely mother-calf pairs and juveniles, various life stages of green, leatherback, and hawksbill sea turtles, juvenile and adult Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat. However, we are not able to estimate the amount of take or area of critical habitat to be affected at this time because we do not know the location, size, or type of MEC/MPPEH. This information will be determined as part of future step-down consultations for specific removal activities proposed by the Navy.

Similarly, we are not able to estimate the area of elkhorn and staghorn coral critical habitat that will be affected by encapsulation and removal of encrusted items from areas containing the PBF at this time. This will also be part of future step-down consultations for specific removal activities once these effects are known.

9 CUMULATIVE EFFECTS

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 C.F.R. §402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

For this consultation, cumulative effects include climate change, fishing, vessel operation and traffic, research activities, coastal and marine development, military cleanup activities, and natural disturbance. With continuing climate change, natural disturbance from storms may increase. Climate change continues to cause increasing prolonged periods of elevated sea surface temperatures, which affects the health of ESA-listed corals in particular. Sea level rise has already been measured in Puerto Rico and is projected to continue. These changes due to climate change could lead to shifts in coastal habitats that could contribute additional MEC/MPPEH items to the marine environment over time.

Fishing and research activities are expected to continue into the foreseeable future. We are not aware of any proposed or anticipated changes in fishing and research activities that would substantially change the impacts of these activities on green, hawksbill, and leatherback sea turtles, Nassau grouper, sperm whales, and ESA-listed corals and elkhorn and staghorn coral designated critical habitat.

Military activities are no longer ongoing but terrestrial cleanup activities that can generate stormwater runoff and associated sediment transport to nearshore waters due to vegetation clearing and demolition operations are expected to continue for some time. Once terrestrial cleanup activities are complete, there could be increases in coastal development and vessel traffic in various locations around Vieques. Ongoing climate change could exacerbate the effects of any increases in land clearing and development as increased storms would lead to more runoff and the transport of land-based pollutants to nearshore waters used by sea turtles, Nassau grouper, and ESA-listed corals.

10 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and critical habitat because of implementing the action. In this section, we add the *Effects of the Action* (Section 8) to the *Environmental Baseline* (Section 8) and the *Cumulative Effects* (Section 9) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the *Status of the Species and Critical Habitat* (Section 6.2).

Some ESA-listed species and designated critical habitat are located within the action area but are not expected to be affected by the action or the effects of the action on these ESA resources were determined to be insignificant or discountable. Some activities evaluated individually were determined to have insignificant or discountable effects and thus to be not likely to adversely affect some ESA-listed species and designated critical habitat (Sections 6.1 and 8.1).

The following discussions separately summarize the probable risks the proposed action poses to sperm whales, green (North and South Atlantic DPSs), hawksbill, and leatherback sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the activities considered further in this Opinion; specifically survey and removal activities involving towing of underwater equipment or MEC/MPPEH, BIPs, nonintentional detonations, encapsulation, removal of encrusted items in coral habitats, use of cast nets and fish traps, in-water structures that present an entrapment hazard, coral tissue sampling, and collection and transport of ESA-listed corals. Up to 5,173 ESA-listed coral colonies will be taken as a result of equipment collisions (estimated as two coral colonies per year dead or damaged), tissue sampling (50 coral colonies over 20 years), and up to the total number (5,173) transplanted, of which 517 would suffer mortality from transplant stress. Bycatch of juvenile sea turtles and Nassau grouper during biological sampling will result in the non-lethal take of 10 juvenile green sea turtles, one juvenile hawksbill sea turtle, and two juvenile Nassau grouper. In-water structures will result in take through entrapment of 103 green sea turtle hatchlings, five of which will suffer mortality as a result of predation, 100 leatherback hatchlings, five of which will suffer mortality as a result of predation, and 202 hawksbill hatchlings, 10 of which will suffer mortality as a result of predation. Additionally, while we discussed the effects of underwater detonations from BIPs or nonintentional detonations in this Opinion, step-down consultations will be required to fully consider the extent and effects of these on sperm whales, sea turtles, Nassau grouper, and elkhorn and staghorn coral critical habitat because take of these animals and damage to critical habitat are expected as a result of the physical effects of underwater detonations and noise generated by detonations depending on the location and size of the MEC/MPPEH. Further, while we discussed the effects of encapsulation and removal of encrusted items on elkhorn and staghorn coral critical habitat, step-down consultations will be required to fully consider the effects of these activities on critical habitat depending on the size of the area to be encapsulated or excavated.

10.1 Jeopardy Analysis

The jeopardy analysis relies upon the regulatory definition of “to jeopardize the continued existence of a listed species,” which is “to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 C.F.R. §402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

Based on our effects analysis, adverse effects to ESA-listed species are likely to result from the action. The following discussions summarize the probable risks that removal activities pose to threatened and endangered species that are likely to be exposed over the 20-year lifetime of the action. These summaries integrate our exposure, response, and risk analyses from Section 8.2.

10.1.1 Sperm Whales

Sperm whales are present seasonally in the action area and are expected to be exposed to noise from underwater detonations associated with BIPs and nonintentional detonations that occur during the winter months when mother-calf pairs and juveniles of the species are likely to be in the action area. The severity of an animal's response to noise associated with underwater detonations will depend on the location of the detonation in relation to the deepwater areas where these animals are more likely to be present during winter months.

Sperm whales are thought to be the most abundant large whale species though there are insufficient data to evaluate trends in abundance and growth rates. The marine mammal stock assessment reports indicate the U.S. Caribbean may contain a separate stock of sperm whales but there are insufficient data to assess the population. There are reports indicating that sperm whales frequent the U.S. Caribbean during their winter migration and there have been sightings of mother-calf pairs and juveniles, as well as a stranding of a juvenile in 2013. Thus, we expect that mother-calf pairs and juveniles are the life stages of sperm whales that may be affected by take in the form of PTS, TTS, or behavioral changes should underwater detonations occur as a result of the proposed removal activities. Take may have short or long-term consequences, depending on the level of noise from detonations to which animals are exposed. This will be discussed further in step-down consultations for removal activities when we know more about where underwater detonations may occur. The anticipated take of a mother-calf pair and/or a juvenile in years when underwater detonations occur could lead to a loss of reproduction at an individual level, but is not expected to have a measurable effect on reproduction at the population level.

The action will not affect the current geographic range of sperm whales and no reduction in the distribution of this species is expected as a result of the action. For this reason, we do not expect the take of individuals to result in population-level consequences to sperm whales.

Because we do not anticipate a significant reduction in numbers or reproduction of this species as a result of the action, particularly removal activities that we determined were likely to result in adverse effects to sperm whales, a reduction in the likelihood of survival for sperm whales is not expected.

The 2010 Recovery Plan (NMFS 2010) for sperm whales identifies recovery criteria geographically across three ocean basins with the following recovery goals:

1. Achieve sufficient and viable populations in all ocean basins.
2. Ensure significant threats are addressed.

No significant changes in population or the extent or magnitude of threats to sperm whales are anticipated as a result of the action. There could be a slight reduction in reproduction, at least in the year when individuals are affected by underwater detonations, should they occur, but this will not have measurable effects on reproduction at the population level. Therefore, we do not anticipate that the action will impede the recovery goals for sperm whales. We conclude that the proposed action will not jeopardize the continued existence of sperm whales.

10.1.2 Sea Turtles

Even if take is non-lethal, individuals may expend more energy fleeing from noise from underwater detonations, suffer hearing impairment, or experience a stress response from being trapped in cast nets and fish traps (if small juvenile green or hawksbill sea turtles), or along in-water structures (if hatchlings). This can result in reduced growth rates, older age to maturity, and lower lifetime fecundity. Nesting females that experience non-lethal take may also have a reduced reproductive output.

Green Sea Turtle, North Atlantic and South Atlantic DPSs

We anticipate that 10 juvenile green sea turtles (from either DPS) in years when cast nets and fish traps are used to perform biological sampling, and 103 green sea turtle hatchlings annually (from either DPS), five of which would suffer mortality as a result of predation due to entrapment in in-water structures seaward of a nesting beach, will be taken as a result of the action. Additional take of adult, juvenile, and hatchling green sea turtles could occur as a result of underwater detonations from BIPs and nonintentional detonations. The severity of an individual animal's response to noise and fragments from detonations will depend on the location and magnitude of the detonation. This take will be discussed further in step-down consultations for removal activities when we know more details about where underwater detonations may occur.

No reduction in the distribution or current geographic range of green sea turtles from either DPS is expected from the anticipated take.

Whether the potential reduction in numbers due to lethal take or due to impacts to reproductive output would appreciably reduce the likelihood of survival of green sea turtles from either DPS depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The North Atlantic DPS is the largest of the 11 green sea turtle DPSs with an estimated abundance of over 167,000 adult females from 73 nesting sites. All major nesting populations demonstrate long-term increases in abundance (Seminoff et al. 2015). The South Atlantic DPS is large, estimated at over 63,000 nesting females, but data availability is poor with 37 of the 51 identified nesting sites not having sufficient data to estimate the number of nesters or trends (Seminoff et al. 2015). While the lack of data is a concern due to increased uncertainty, the overall trend of the South Atlantic DPS was not considered to be a major concern because some of the largest nesting beaches such as Ascension Island and Aves Island in Venezuela and Galibi in Suriname appear to be increasing with others (Trindade, Brazil; Atol das Rocas, Brazil; Poilão and the rest of Guinea-Bissau) appearing to be stable. In the U.S., nesting of green sea turtles occurs in the South Atlantic DPS on beaches of the U.S. Virgin Islands, primarily on Buck Island and Sandy Beach, St. Croix, although there are not enough data to establish a trend. Because Vieques is considered part of the Virgin Islands due to its geographic location, it is possible that green sea turtles nesting on the island are from either DPS.

We believe the action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of green sea turtles from either DPS in the wild. Also the potential mortality of various life stages of green sea turtles may occur as a result of the action, particularly noise effects associated with underwater detonations, and would result in a reduction in absolute population numbers, the population of green sea turtles in either DPS would not be appreciably affected. Likewise, the reduction in reproduction that could occur as a result of mortality of individuals or decreased growth rates of earlier life stages would not appreciably affect reproductive output in the North or South Atlantic. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals from successful reproduction of sea turtles that are not taken as a result of the action. Because the abundance trend information for green sea turtles is increasing (North Atlantic DPS) or stable (South Atlantic DPS), we believe the anticipated takes attributed to the action will not have any measurable effect on the trend for either DPS.

The Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) lists the following recovery objective for a period of 25 continuous years that is relevant to the impacts of the proposed action:

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

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern United States and U.S. Caribbean. From 2000 – 2006, sea turtle surveys in Culebra resulted in the capture of 553 green sea turtles and all were juveniles or subadults based on size and testosterone levels thus suggesting Culebra is an important developmental habitat (Diez et al. 2007). Green sea turtles are frequently sighted around Vieques but no in-water sea turtle survey data are available.

The potential take of 103 green hatchlings per year with five of these expected to suffer predation associated with entrapment in in-water structures installed seaward of a nesting beach and 10 juvenile green sea turtles is not likely to reduce population numbers over time given current population sizes and expected recruitment. Similarly, while we cannot estimate the exact numbers of take of adult, juvenile, and hatchling green sea turtles that may occur as a result of underwater detonations, we do not expect a significant reduction in population numbers due to the stressors associated with these activities. Thus, the action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of green sea turtles' recovery in the wild. We conclude that the proposed action will not jeopardize the continued existence of green sea turtles.

Leatherback Sea Turtle

We believe there is the potential for the take of 100 leatherback hatchlings annually due to entrapment in in-water structures such as floating barriers installed seaward of a nesting beach (assumed to have an average of three leatherback nests). We also anticipate that five of these animals will suffer mortality due to predation. Additional take of adult female and hatchling leatherback sea turtles could occur as a result of underwater detonations from BIPs and nonintentional detonations. The severity of individual animal's responses to noise and fragments from detonations will depend on the location and magnitude of the detonation. This take will be discussed further in step-down consultations for removal activities when we know more details about where underwater detonations may occur.

Given these sea turtles generally have large ranges in which they disperse, no reduction in the distribution or current geographic range of leatherback sea turtles is expected as a result of the proposed action.

The anticipated take of hatchlings is anticipated annually at 5 percent due to predation. The take of hatchlings would result in a slight reduction in absolute population numbers and an associated slight reduction in reproduction. Take of adult female leatherback sea turtles and additional take of hatchlings could occur as a result of underwater detonations. This take would result in PTS, TTS, or behavioral responses and could result in a loss of individuals, which would also mean a loss of reproduction. It is not likely this reduction would appreciably reduce the likelihood of survival of leatherback sea turtles. Nesting trends for the Florida and Northern Caribbean populations, including the largest nesting population in the Southern Caribbean, are all either stable or increasing. Nesting by leatherbacks is reported on various beaches in the action area that would not be affected by the installation of in-water structures seaward of nesting beaches, and underwater detonations are expected to be extremely rare, if they occur at all. Thus, we believe the proposed action is not likely to have any measurable effect on overall population trends.

Because we do not anticipate a significant reduction in numbers or reproduction of this species as a result of the action, a reduction in the likelihood of survival for leatherback sea turtles is not expected.

The Atlantic Recovery Plan for the U.S. population of the leatherback sea turtle (NMFS and USFWS 1992) listed the following relevant recovery objective:

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

Between 1978 – 2005, leatherback nesting increased in Puerto Rico from a minimum of 9 nests recorded in 1978 to 469 – 882 nests recorded each year from 2000 – 2005. The annual rate of increase in nesting was estimated to be 1.1 with a growth rate interval between 1.04 – 1.12, using nesting numbers from 1978 – 2005 (USFWS and NMFS 2007b).

The potential take of 100 leatherback hatchlings per year with five of these expected to suffer predation associated with entrapment in in-water structures installed seaward of a nesting beach is not likely to reduce population numbers over time given current population sizes and expected recruitment. Similarly, while we cannot estimate the exact numbers of take of adult female and hatchling leatherback sea turtles that may occur as a result of underwater detonations, we do not expect a significant reduction in population numbers due to the stressors associated with these activities. Thus, the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. We conclude that the proposed action will not jeopardize the continued existence of leatherback sea turtles.

Hawksbill Sea Turtle

We believe there is the potential for take of one juvenile hawksbill sea turtle due to the use of cast nets and fish traps in years when biological sampling takes place and 202 hawksbill hatchlings annually due to entrapment in in-water structures installed seaward of a nesting beach of which 10 would suffer mortality from predation. Additional take of adult, juvenile, and hatchling hawksbill sea turtles could occur as a result of underwater detonations from BIPs and nonintentional detonations. The severity of an individual animal's response to noise and fragments from detonations will depend on the location and magnitude of the detonation. This take will be discussed further in step-down consultations for removal activities when more details are known about where underwater detonations may occur.

No reductions in the distribution or current geographic range of hawksbill sea turtles is expected from the anticipated take.

Whether the potential reduction in numbers due to lethal take or due to impacts to reproductive output would appreciably reduce the likelihood of survival of hawksbill sea turtles depends on the probably effect the changes in numbers and reproduction would have relative to current population sizes and trends. 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 are currently the primary information source for evaluating trends in abundance. Mortimer and Donnelly (2008) found that for nesting populations in the Atlantic (especially in the Insular Caribbean and Western Caribbean Mainland), 9 of the 10 sites with recent data (within the past 20 years from approximately 1988 to 2008) that show nesting increases in the Caribbean. With increasing nesting trends in the Caribbean, we believe the losses expected due to the action will be replaced due to increased nest production. Therefore, we believe the reduction in numbers and reproduction will not appreciably reduce the survival of hawksbill sea turtles in the wild.

The Recovery Plan for the population of hawksbill sea turtle (NMFS and USFWS 1993) listed the following relevant recovery objectives over a continuous 25-year period:

- The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests at five index beaches, including Mona Island (Puerto Rico) and Buck Island Reef National Monument (St. Croix).
- The 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, U.S. Virgin Islands, and Florida.

Of the hawksbill sea turtle rookeries regularly monitored – Jumby Bay (Antigua/Barbuda), Barbados, Mona Island (Puerto Rico), and Buck Island Reef National Monument (St. Croix)-- all show increasing trends in the annual number of nests (USFWS and NMFS 2007a). In-water research projects at Mona Island, Buck Island, and the Marquesas, Florida, which involve the observation and capture of juvenile hawksbill sea turtles have been conducted (USFWS and NMFS 2007a). Although there are over 15 years of data for the Mona Island project, abundance indices have not yet been incorporated into a rigorous analysis or a published trend assessment. The time series for the Marquesas project is not long enough to detect a trend.

The potential take of 202 hawksbill hatchlings per year with 10 of these expected to suffer predation associated with entrapment in in-water structures installed seaward of a nesting beach, and one juvenile hawksbill sea turtle is not likely to reduce population numbers over time given current population sizes and expected recruitment. Similarly, while we cannot estimate the exact numbers of take of adult, juvenile, and hatchling hawksbill sea turtles that may occur as a result of underwater detonations, we do not expect a significant reduction in population numbers due to the stressors associated with these activities. Thus, the action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild. We conclude that the proposed action will not jeopardize the continued existence of hawksbill sea turtles.

10.1.3 Nassau Grouper

We believe there is the potential for lethal and non-lethal take of two juvenile Nassau grouper due to the use of cast nets and fish traps in years when biological sampling takes place. Additional lethal and non-lethal take of adult and juvenile Nassau grouper could occur as a result of underwater detonations from BIPs and nonintentional detonations. The severity of an individual animal's response to noise and fragments from detonations will depend on the location and magnitude of the detonation. This take will be discussed further in step-down consultations for removal activities when more details are known about where underwater detonations may occur.

No reductions in the distribution or current geographic range of Nassau grouper is expected from the anticipated take.

Whether the potential reduction in numbers due to lethal take or due to impacts to reproductive output would appreciably reduce the likelihood of survival of Nassau grouper depends on the probably effect the changes in numbers and reproduction would have relative to current population sizes and trends. There are currently no reliable estimates of population abundance and trends but Sadovy et al. (2018) estimated the overall population at 3,000. Fishing of Nassau grouper has been prohibited in the U.S. Caribbean and there is some evidence that multispecies SPAGS now include Nassau grouper in increasing numbers (Kadison et al. 2009; Schärer et al. 2009). There are no estimates of juvenile abundance but it would be expected to increase as more adults spawn annually. Lethal take of Nassau grouper as a result of the action would lead to reductions in reproductive output and non-lethal take could also affect reproductive output. Juveniles may be captured during biological sampling and those that suffer mortality would never reproduce while those that suffer non-lethal take could have delayed growth . Given the limited amount of biological sampling and the unlikelihood of underwater detonations, as well as the large habitat areas available to juvenile and adult Nassau grouper where no removal activities are likely to occur, we believe the number of individuals affected by the action is likely to be a very small percentage of the actual population in the action area. Therefore, we believe the reduction in numbers and reproduction will not appreciably reduce the survival of Nassau grouper in the wild.

A recovery plan is not available for Nassau grouper but NMFS has developed a recovery outline for this species (available at https://www.fisheries.noaa.gov/resource/document/nassau-grouper-recovery-outline). The outline serves as an interim guidance document to direct recovery efforts, including recovery planning, until a full recovery plan is developed and approved. The Summary Assessment in the recovery outline concludes that Nassau grouper are now at a very small fraction of their historic abundance. Therefore, conservation and recovery of Nassau grouper requires a two-pronged approach focusing on: 1) reproduction and recruitment as essential with spawning aggregations continuing to function throughout the range to provide larvae, and 2) ensuring appropriate habitat is available for settlement and growth across the Caribbean Sea. The major threat to Nassau grouper is fishing.

To determine if the action will appreciably reduce the likelihood of recovery for Nassau grouper, we assess the effects of the proposed action in the context of our knowledge of the status of the species, its environmental baseline, the extinction risk analyses in the listing rule, and the information in the recovery outline. The final listing rule identified the species' abundance, life history characteristics, and threat vulnerabilities as characteristics that increase extinction risk. Its low abundance compared to its historic population estimates exacerbate its vulnerability to extinction. Nassau grouper are present in the action area based on survey data but there are no estimates of the number of these animals present. The proposed action will not affect the species' life history characteristics or increase the magnitude of the species' vulnerability to fishing, although fishing for this species in the action area and all of the U.S. Caribbean waters is prohibited. The action will cause a small decrease in reproductive potential and will affect habitat used by the species, particularly juvenile habitat in shallow water, through removal actions. The area affected is a small portion of the species' range and the number of individuals that may be affected by the proposed action is likely a small portion of the population of Nassau grouper present in the action area.

The potential take of two juvenile Nassau grouper in years when biological sampling occurs is not likely to reduce population numbers over time given current population sizes and expected recruitment. Similarly, while we cannot estimate the exact numbers of take of adult and juvenile Nassau grouper that may occur as a result of underwater detonations, we do not expect a significant reduction in population numbers due to the stressors associated with these activities. Thus, the action is not likely to impede the recovery priorities identified for Nassau grouper and will not result in an appreciable reduction in the likelihood of Nassau grouper's recovery in the wild. We conclude that the action will not jeopardize the continued existence of Nassau grouper.

10.1.4 ESA-Listed Corals

As discussed in this Opinion, 5,173 ESA-listed coral colonies are expected to be adversely affected by the action over the 20-year lifetime of the action. We are unable to separate this estimate into the numbers of colonies of each listed coral species that may be affected. However, data from surveys conducted in the action area indicate that boulder and mountainous star corals and staghorn coral are the most abundance species. Pillar and rough cactus corals are naturally rare.

We estimate that two ESA-listed coral colonies will be taken annually due to collisions with towed equipment and towed MEC/MPPEH; 50 colonies will be taken due to tissue sampling over the 20-year timeline; and all of the 5,173 colonies could be taken due to collection and transplant (assuming they suffered only damage or partial mortality from other activities resulting in take), of which 517 would be expected to suffer mortality as a result of handling and transplant stress in the year when this occurs. Additional take of ESA-listed coral colonies could occur as a result of underwater detonations from BIPs or nonintentional detonation depending on the location and magnitude of the detonation. This take will be discussed further in step-down consultations for removal activities where more details are known about underwater detonations may occur.

Elkhorn and Staghorn Corals

The abundance of elkhorn and staghorn coral is a fraction of what it was before the mass mortality in the 1970s and 80s and recent population models forecast the extirpation of elkhorn coral from some locations over the foreseeable future, including a site in Vieques that was included in the Jackson et al. (2014b) report. The presence of staghorn coral on reefs throughout its range has continued to decrease. Elkhorn corals occupy habitats from back reef environments to turbulent water on the fore reef, reef crest, and shallow spur-and-groove zone, which moderates the species' vulnerability to extinction although many of the reef environments it occupies will experience highly variable thermal regimes and ocean chemistry due to climate change. Staghorn corals occupy a broad range of depths and multiple, heterogeneous habitat types, including deeper waters, which moderates the species' vulnerability to extinction over the foreseeable future. Elkhorn coral abundance is at least hundreds of thousands of colonies but likely to decrease in the future with increasing threats. Staghorn coral abundance is at least tens of millions of colonies but likely to decrease in the future with increasing threats.

No reductions in the distribution or geographic range of elkhorn and staghorn coral are expected to occur as a result of the action.

The action is expected to result in the lethal and non-lethal take of elkhorn and staghorn coral colonies. It is not possible for us to estimate the total numbers of colonies of each species that will be taken but these are likely to be a fraction of the total present in the action area. The loss of elkhorn and staghorn coral colonies will result in a reduction in absolute population numbers of these species in the action area. The loss or temporary removal from the reproductive pool of sexually mature colonies due to responses such as transplant stress will also result in the loss of reproductive potential.

Despite the potential loss of elkhorn and staghorn coral colonies and reproductive potential, the area to be affected is part of an extensive reef system between mainland Puerto Rico and the Virgin Islands. Whether the expected reduction in future reproduction of elkhorn and staghorn corals would appreciably reduce their likelihood of survival depends on the probable effect the changes in reproduction would have relative to the current population levels and trends. Based on best available population estimates, there are at least hundreds of thousands of elkhorn coral colonies and at least tens of millions of staghorn coral colonies present in the Florida Keys and St. Croix, USVI. Absolute abundance is higher than estimates from these locations alone given the presence of these species in many other locations throughout their range, including around Puerto Rico. In the status of the species section, we concluded there has been a significant decline in elkhorn coral throughout its range with recent population stability at low percent cover and that local extirpations are possible. We conclude that staghorn coral has declined throughout its range as well.

Elkhorn coral has low sexual recruitment rates, meaning that genetic heterogeneity is low. However, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating its vulnerability to extinction. Also, given elkhorn coral's estimated abundance, the loss of reproductive potential represented by take of elkhorn colonies due to the proposed action over 20 years will not measurably impact the species' abundance in Puerto Rico or throughout the species' range. Therefore, we believe the loss of elkhorn coral colonies and reproductive potential due to the action will not appreciably reduce elkhorn coral's ability to survive in the wild.

Staghorn corals occur throughout the Caribbean Basin and the corals in the action area account for a very small portion of the total numbers of or area occupied by staghorn coral. The species' absolute abundance is at least tens of millions of colonies, based on estimates from only two locations. Impacts to the species' areal coverage would also likely be undetectable on a

Caribbean-wide scale. Therefore, we believe the loss of staghorn coral colonies and reproductive potential due to the action will not appreciably reduce staghorn coral's ability to survive in the wild.

The recovery plan for elkhorn and staghorn corals outlines a recovery strategy for the species:

> "Elkhorn and staghorn coral populations should be large enough so that successfully reproducing individuals comprise numerous populations across the historical ranges of these species and are large enough to protect their genetic diversity and maintain their ecosystem functions. Threats to these species and their habitat must be sufficiently abated to ensure a high probability of survival into the future" (NMFS 2015b).

The recovery plan established three recovery criteria associated with the objective of ensuring population viability and seven recovery criteria associated with the objective of eliminating or sufficiently abating global, regional, and local threats that contribute to species' status. The best available information indicates that all recovery objectives must be met for elkhorn and staghorn corals to achieve recovery. The most relevant criteria to the impacts expected form the proposed action include:

Objective 1: Ensure Population Viability

Criterion 1: Abundance

Elkhorn coral: Thickets are present throughout approximately 10% of consolidated reef habitat in 1 – 5 m water depth within the forereef zone. Thickets are defined as either a) colonies $\geq$ 1 m diameter in size at a density of 0.25 colonies per m^2 or b) live elkhorn coral benthic cover of approximately 60 percent. Populations with these characteristics should be present throughout the range and maintained for 20 years.

Staghorn coral: Thickets are present throughout approximately 5% of consolidated reef habitat in 5 – 20 m water depth within the forereef zone. Thickets are defined as either a) colonies $\geq$ 0.5 m diameter in size at a density of 1 colony per m^2 or b) live staghorn coral benthic cover of approximately 25 percent. Populations with these characteristics should be present throughout the range and maintained for 20 years.

Objective 2: Eliminate or Sufficiently Abate Global, Regional, and Local Threats

Criterion 6: Loss of Recruitment Habitat

Abundance (Criterion 1 above) addresses the threat of Loss of Recruitment Habitat because the criterion specifies the amount of habitat occupied by the 2 species. If Criterion 1 is met, then this threat is sufficiently abated; or

Throughout the ranges of these 2 species, at least 40 percent of the consolidated reef substrate in 1 – 20 m depth within the forereef remains free of sediment and macroalgal cover as measured on a broad reef to regional spatial scale.

In terms of the recovery objectives, the action is not expected to reduce the overall abundance of elkhorn and staghorn corals in the action area. In terms of Recovery Objective 1 and based on information in the BA and in supplemental images and videos provided by the Navy for the consultation, elkhorn or staghorn coral thickets are not present in the majority of areas where suspected MEC/MPPEH items are present and may be subject to removal. Thus, we do not expect the abundance objective to be affected. Although we do anticipate some effects to elkhorn and staghorn coral critical habitat, we expect recruitment habitat to remain in the action area within the percentage established to meet Recovery Objective 2. Therefore, even with the loss of a small area of critical habitat from the action area due to the construction and operation of the project, we do not believe there will be an appreciable reduction in the likelihood of recovery in the wild for elkhorn and staghorn corals. We conclude that the proposed action will not jeopardize the continued existence of elkhorn and staghorn corals.

Pillar Coral

We do not have precise population estimates for the species. The listing rule (79 FR 53852, September 10, 2014) notes that there are at least tens of thousands of colonies in the Florida Keys, although many of these have suffered full or partial mortality due to a tissue loss disease documented in 2017 (see Section 6.2.5.3). The species is naturally uncommon to rare and population estimates for the Caribbean are not available. Pillar coral is distributed throughout most of the greater Caribbean in reef environments between 1 – 25 m in depth but the low coral cover of this species makes it difficult to extrapolate monitoring data in order to determine trends in abundance. Based on information in our project files from other sites in the U.S. Caribbean, pillar coral appears to be more common around Puerto Rico and USVI in general than in South Florida (NOAA, NCRMP).

No reductions in the distribution or geographic range of pillar coral is expected to occur as a result of the proposed action.

We find that the anticipated lethal and non-lethal take of pillar coral colonies associated with the action will result in a reduction in numbers of this species. Pillar corals are most likely to be affected by underwater detonations and collection and transplant. Transplanted corals are likely to suffer partial tissue mortality and bleaching and 10 percent of them are likely to die as a result of the stress of transplantation. The pillar coral colonies affected by the action are expected to be a fraction of those present in the action area.

The reduction in numbers of pillar corals in the action area is expected to result in a loss of reproductive potential over the 20-year lifetime of the proposed action. Despite the potential loss of reproductive potential, the action area represents a very small portion of the species' range and, based on information from coral surveys in Puerto Rico and USVI, pillar corals may be more common in the U.S. Caribbean than in other areas within the species' range. Despite the reduction in reproductive potential, we do not believe there will be long-term damage to the species' ability to sexually reproduce as a result of the action. Therefore, although we believe the project will lead to a loss of reproductive potential related to mortality of colonies that are sexually mature and the temporary loss of reproductive potential due to stressors such as transplantation, we do not anticipate that this would represent a detectable reduction in the long-term reproduction of pillar coral in the action area. We believe the lethal and non-lethal take of pillar coral colonies in the action area over 20 years will not have any measurable effect on the overall population and will not appreciably reduce the species' likelihood of survival in the wild.

A recovery plan is not available for pillar corals but NMFS has developed a recovery outline for this species (available at https://www.fisheries.noaa.gov/resource/document/5-caribbean-coral-species-recovery-outline). The outline serves as an interim guidance document to direct recovery efforts, including recovery planning, until a full recovery plan is developed and approved. The Summary Assessment in the recovery outline concludes that population trends for pillar corals are unknown. Therefore, recovery will depend on successful sexual reproduction and reducing mortality of extant populations. The key challenges will be moderating the impacts of ocean warming associated with climate change and decreasing susceptibility to disease, which may be furthered through reduction of local stressors. The recovery of the species will require an ecosystem approach including habitat protection measures, a reduction in threats caused by human activity, additional research, and time. The recovery vision for the species concludes that it should be present across its historic range, with populations large enough and genetically diverse enough to support successful reproduction and recovery from mortality events and dense enough to maintain ecosystem function.

To determine if the proposed action will appreciably reduce the likelihood of recovery for pillar corals, we assess the effects of the proposed action in the context of our knowledge of the status of the species, its environmental baseline, the extinction risk analyses in the listing rule, and the information in the recovery outline. The final listing rule identified the species' abundance, life history characteristics, depth distribution, and threat vulnerabilities as characteristics that increase extinction risk. Its low abundance, combined with its geographic location in shallow waters, exacerbate its vulnerability to extinction. Pillar corals are present in the action area in waters up to approximately 60 ft based on NCRMP data. The action will not affect the species' life history characteristics or increase the magnitude of the species' vulnerability to climate change threats such as ocean warming. The action will cause a small decrease in reproductive potential and will affect habitat for the species through removal actions. The area affected is a small portion of the species' range and the number of colonies that may be affected by the action is likely a small portion of the pillar coral colonies present in the action area. Therefore, we believe that the impacts to pillar corals resulting from the action will not increase the magnitude of the threats that led to the listing of the species as threatened to levels that will appreciably reduce this species' likelihood of recovery in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of pillar corals in the wild.

Rough Cactus Coral

Rough cactus coral is reported in the Caribbean and western Atlantic with the exceptions of the Flower Garden Banks, Bermuda, Brazil, and the southeast U.S. north of South Florida. Rough cactus coral is one of the least common coral species observed when it is present.

No reductions in the distribution or geographic range of rough cactus coral is expected to occur as a result of the action.

We find that the anticipated lethal and non-lethal take of rough cactus coral colonies associated with the action will result in a reduction in numbers of this species. Rough cactus corals are most likely to be affected by underwater detonations and collection and transplant. Transplanted corals are likely to suffer partial tissue mortality and bleaching and 10 percent of them are likely to die as a result of the stress of transplantation. The reduction in numbers of rough cactus corals in the action area is also expected to result in a loss of reproductive potential, both permanent (due to mortality) and temporary (due to things like transplant stress). Whether the expected reduction in reproduction of rough cactus corals will appreciably reduce its likelihood of survival depends on the probable effects the changes in reproduction would have relative to the current population levels and trends.

Low encounter rate and low percent cover, as well as a tendency to identify *Mycetophyllia* only to genus in surveys, make it difficult to discern population trends from monitoring data. However, reported losses of rough cactus corals from monitoring stations in the Florida Keys and Dry Tortugas indicate populations have declined in these areas. Based on the declines in Florida, the listing rule concluded that rough cactus coral has likely decline throughout its range. The population of the species is estimated as at least hundreds of thousands based on estimates from two locations, meaning absolute abundance is higher because the species occurs in many other locations throughout its range. Rough cactus coral is a hermaphroditic brooding spawner with very low recruitment. The species has been classified as a generalist, weedy, competitive, and stress-tolerant (Darling et al. 2012), meaning that it is expected to be more resistant to environmental stress than other listed coral species. NCRMP surveys documented the species in the action area in waters approximately 60 ft in depth or deeper. We believe the loss of rough cactus corals as a result of the action will not have a measurable effect on the overall population and is not likely to appreciably reduce the species' likelihood of survival in the wild.

A recovery plan is not available for pillar corals but NMFS has developed a recovery outline for this species (available at https://www.fisheries.noaa.gov/resource/document/5-caribbean-coral-species-recovery-outline). The outline serves as an interim guidance document to direct recovery efforts, including recovery planning, until a full recovery plan is developed and approved. The Summary Assessment in the recovery outline concludes that population trends for rough cactus corals are unknown but the species does appear to have experienced a decline in Florida. Therefore, recovery will depend on successful sexual reproduction and reducing mortality of extant populations. The key challenges will be moderating the impacts of ocean warming associated with climate change and decreasing susceptibility to disease, which may be furthered through reduction of local stressors. The recovery of the species will require an ecosystem approach including habitat protection measures, a reduction in threats caused by human activity, additional research, and time. The recovery vision for the species concludes that it should be present across its historic range, with populations large enough and genetically diverse enough to support successful reproduction and recovery from mortality events and dense enough to maintain ecosystem function.

To determine if the action will appreciably reduce the likelihood of recovery for rough cactus corals, we assess the effects of the action in the context of our knowledge of the status of the species, its environmental baseline, the extinction risk analyses in the listing rule, and the information in the recovery outline. The final listing rule identified the species' abundance, life history characteristics, and threat vulnerabilities as characteristics that increase extinction risk. Its low abundance, combined with its geographic location, exacerbate its vulnerability to extinction. Pillar corals are present in the action area in waters up to approximately 60 ft based on NCRMP data. The action will not affect the species' life history characteristics or increase the magnitude of the species' vulnerability to climate change threats such as ocean warming. The action will cause a small decrease in reproductive potential and will affect habitat for the species through removal actions. The area affected is a small portion of the species' range and the number of colonies that may be affected by the action is likely a small portion of the rough cactus coral colonies present in the action area. Therefore, we believe that the impacts to rough cactus corals resulting from the action will not increase the magnitude of the threats that led to the listing of the species as threatened to levels that will appreciably reduce this species' likelihood of recovery in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of rough cactus corals in the wild.

Lobed Star, Boulder Star, and Mountainous Star Corals

The star coral complex has historically been dominant on coral reefs in the Caribbean and has been the major reef builder in the Caribbean since elkhorn and staghorn corals began to decline in abundance. However, multiple reports from various countries indicate the populations of corals from the star coral complex are in decline, including the U.S. (Florida, USVI, and Puerto Rico), Curaçao, Belize, and Colombia. As for other areas in the Caribbean, corals from the star coral complex dominate in the action area.

No reductions in distribution or the geographic range of lobed star, boulder star, and mountainous star corals is expected as a result of the action.

We conclude that the action will result in a reduction in numbers of these species. It is not possible for us to estimate the total numbers of colonies of each species that will be taken but these are likely to be a fraction of the total present in the action area given the dominance of these hard coral species in the action area and throughout the Caribbean. The loss of lobed star, boulder star, and mountainous star coral colonies will result in a reduction in absolute population numbers of these species in the action area. The loss or temporary removal from the reproductive pool of sexually mature colonies due to responses such as transplant stress will also result in the loss of reproductive potential. Despite the anticipated loss of reproductive potential due to the action, we do not believe sexually reproductive individuals of these species in the action area would be affected to a degree that will cause short or long-term damage to the species' ability to sexually reproduce.

Whether the reduction in numbers and reproduction of these species would appreciably reduce their likelihoods of survival in the wild depends on the probable effects these changes would have relative to current population status and trends. Information on the distribution and cover of lobed star, boulder star, and mountainous star corals around Puerto Rico indicate that they are dominant on mesophotic reefs in Puerto Rico and USVI at depths up to 90 m, although boulder star coral tends to be the most dominant species at greater depths and lobed star coral in shallow depths. Species from this complex often make up the largest proportion of coral cover on Caribbean reefs, including survey sites on several reefs in Puerto Rico despite impacts from the 1998 and 2005 mass bleaching events and 2017 hurricanes. Lobed star coral has been estimated as having an absolute abundance of at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined. Mountainous star coral's absolute population abundance has been estimated as at least tens of millions of colonies in each of several locations, including the Florida Keys, Dry Tortugas, and USVI. Boulder star corals' absolute population abundance has been estimated as at least tens of millions of colonies in the Dry Tortugas and USVI. Therefore, we believe the loss of colonies and reproductive potential due to the action will not appreciably reduce the likelihood of survival in the wild of lobed star, mountainous star, and boulder star corals.

As stated previously for the other species that were listed in September 2014 that will also be affected by the action, there is no recovery plan for these species. However, the recovery plan developed by NMFS (available at https://www.fisheries.noaa.gov/resource/document/5-caribbean-coral-species-recovery-outline) is meant to serve as interim guidance to direct recovery efforts and planning until a full recovery plan is finalized. The Summary Assessment in the recovery outline concludes that overall, available data indicate *Orbicella* coral populations are on the decline and that recovery will depend on successful reproduction and reducing mortality of extant populations. The key challenges will be moderating the impacts of ocean warming associated with climate change and decreasing susceptibility to disease which may be furthered through a reduction of local stressors. The recovery vision statement in the outline states that populations of lobed star, mountainous star, and boulder star corals should be present across their historic ranges with populations large enough and genetically diverse enough to maintain ecosystem function. Given that many of the important threats to the recovery of these species are not directly manageable, the recovery strategy must pursue actions both in the short and long-term to address both global and local threats. The initial focus of the recovery action plan will be to protect extant populations and the species' habitat through reduction of threats. Specific actions identified for early in the recovery process are reducing locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, and over-fishing).

These species' life history characteristics of large colony size and long life span have enabled them to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. The buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes. The action will not affect these life history vulnerabilities or increase the species' vulnerability to ocean warming, disease, nutrient enrichment, or acidification. The action will cause a small decrease in reproductive potential and will affect habitat for the species through removal actions. The area affected is a small portion of the species' range and the number of colonies of each species that may be affected by the action is likely a small portion of the lobed star, boulder star, and mountainous star coral colonies present in the action area. Therefore, we believe that the impacts to lobed star, mountainous star, and boulder star corals resulting from the action will not increase the magnitude of the threats that led to the listing of these species as threatened to levels that will appreciably reduce these species' likelihood of recovery in the wild. We conclude the action is not likely to jeopardize the continued existence of lobed star, mountainous star, and boulder star corals.

10.2 Critical Habitat Destruction/Adverse Modification Analysis

When determining the potential impacts to critical habitat for this Opinion, NMFS relies on the regulatory definition of "destruction or adverse modification" of critical habitat from the revised regulations issued by NMFS and USFWS (84 FR 45016) on August 27, 2019. Under the revised regulations, destruction or adverse modification means a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species.

Ultimately, we seek to determine if, with the implementation of the action, critical habitat would remain functional (or retain the current ability for the PBF to become functional) to serve the intended conservation role for the species. This analysis takes into account the geographic and temporal scope of the 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 must take into account any changes in amount, distribution, or characters of the critical habitat that will be required over time to support the successful recovery of the species.

Within the Puerto Rico elkhorn and staghorn coral critical habitat marine unit, approximately 292 mi^2 (756 km^2) are likely to contain the essential element of ESA-designated elkhorn and staghorn coral critical habitat, based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOAA's NOS Biogeography Program in 2000 (Kendall et al. 2001). The key objective for the conservation and recovery of Atlantic acroporid corals that forms the basis for the critical habitat designation is the facilitation of an increase in the incidence of sexual and asexual reproduction. Recovery cannot occur without protecting the PBF of quality and quantity of suitable substrate because it affects their reproductive success. As noted in the rule designating acroporid coral critical habitat (73 FR 72210, November 26, 2008), the loss of suitable habitat is one of the greatest threats to the recovery of listed elkhorn and staghorn coral populations. Man-made stressors have the greatest impact on habitat quality for listed elkhorn and staghorn corals.

Therefore, the key conservation objective of designated elkhorn and staghorn coral critical habitat is to increase the potential for successful sexual and asexual reproduction, which in turn facilitates increase in the species' abundance, distribution, and genetic diversity. To this end, our analysis seeks to determine whether or not the action is likely to destroy or adversely modify designated critical habitat in the context of the Status of Elkhorn and Staghorn Coral Critical Habitat (Section 6.2.6), the Environmental Baseline (Section 7), the Effects of the Action (Section 8), and Cumulative Effects (Section 9).

The essential feature of critical habitat for elkhorn and staghorn coral is substrate of adequate quantity and quality to allow for settlement and growth where adequate quality refers to the need for hard substrate to be free of high macroalgal growth and sediment cover as these impede the settlement and growth of elkhorn and staghorn corals. Thus, we need to assess whether the potential loss of or damage to critical habitat areas due to underwater detonations during BIPs or nonintentional detonations, encapsulation of MEC/MPPEH, and removal of items encrusted in hard substrate rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Specifically, whether these removal activities will result in diminished function of the PBF of elkhorn and staghorn coral critical habitat such that settlement and growth of sexual and asexual recruits are impaired, also affecting the recovery criteria for elkhorn and staghorn corals.

Our analysis indicates that some removal activities are likely to have permanent effects to small areas of coral critical habitat, such as the installation of anchor pins in hard substrate, underwater detonations from BIPs and nonintentional detonations, encapsulation of MEC/MPPEH, and removal of items encrusted in hard substrate. Each anchor pin has a footprint of 28 in^2, meaning the installation of these pins as anchors for in-water structures such as marker buoys would be minimal. Encapsulation and removal of encrusted MEC/MPPEH in hard substrate in areas with the PBF for coral critical habitat would have a larger footprint, depending on the site of the munitions item or items. However, given the size of the majority of items identified to date during the WAA and other surveys conducted by the Navy, we do not anticipate impact footprints larger than several square feet. In addition, these removal methods have not been used and items determined to be inert that are encrusted in hard substrate are more likely to be left in place with no intervention than encapsulated or broken out of the substrate. Encapsulation is likely to be an option only in cases when MEC/MPPEH are encrusted in hard substrate, likely to present an explosive hazard, and likely to be too unstable to be removed from the substrate without increasing the threat of nonintentional detonation. For this reason, removal of encrusted items is unlikely because any items believed to present an explosive hazard would be left in place rather than trying to chisel these from the substrate due to the increased probability of a nonintentional detonation during removal. BIPs are a removal method that is not likely to be employed but could have a larger habitat impact depending on the location, size, and amount of explosive material both in the munitions item and used to detonate it. Nonintentional detonations could have larger footprints than BIPS if controls are not in place to minimize the magnitude of the blast, should one occur while MEC/MPPEH is being removed from the substrate and towed to a terrestrial disposal location. The actual area of impacts to coral critical habitat from removal activities that may include encapsulation, removal of encrusted items, BIPs, and nonintentional detonations, and from installation of anchor pins will be determined as part of step-down consultations.

Impacts to coral critical habitat from anchor pins and removal activities are expected to be localized and are not expected to result in the loss or degradation of large areas containing the PBF of elkhorn and staghorn coral critical habitat. We base this on the current presence of elkhorn and staghorn corals in areas containing the essential feature within UXO 16, the larger action area, and the Puerto Rico critical habitat unit. The WAA found 5,198 ac of coral habitats within UXO 16, much of which is likely to contain the PBF for elkhorn and staghorn coral critical habitat. Additionally, the underwater survey activities conducted in UXO 16 to date have identified thousands of potential MEC/MPPEH items, many of which have already been removed from non-coral habitats, and from coral habitats if they were resting on the surface with no ESA-listed corals colonizing them and/or no ESA-listed coral colonies within 15 ft of the items with no nonintentional detonations. Some of these removal activities have included remote lifting and towing of items that were suspected to present an explosive hazard with no incident. The majority of items that may be MEC/MPPEH are on the surface based on information in the BA, making removal with little to no habitat damage likely. Therefore, we do not expect the effects of the action to appreciably diminish the overall value of the designated critical habitat for the conservation of elkhorn and staghorn corals in the action area. We conclude that the proposed action will not result in the destruction or adverse modification of elkhorn and staghorn coral critical habitat in the Puerto Rico unit.

11 Conclusion

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of sperm whales, green (North Atlantic and South Atlantic DPSs) sea turtles, leatherback sea turtles, hawksbill sea turtles, Nassau grouper, elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, and boulder star coral, or to result in the destruction or adverse modification of elkhorn and staghorn coral critical habitat.

It is also NMFS biological opinion that the action is not likely to adversely affect the following ESA-listed species: fin whale, sei whale, blue whale, giant manta ray, oceanic whitetip shark, scalloped hammerhead shark (Northwest and Western Central Atlantic DPS), and loggerhead sea turtle (Northwest Atlantic Ocean DPS).

12 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations 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 to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering.

Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement.

12.1 Amount or Extent of 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)). The amount of take represents the number of individuals that are expected to be taken by actions while the extent of take specifies the impact, i.e., the amount or extent of such incidental taking on the species, which may be used if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (see 80 FR 26832).

We anticipate the action associated with the investigation and removal of MEC/MPPEH from UXO 16 around Vieques is reasonably likely to result in the incidental take of ESA-listed species by death, injury, or harassment. Specifically, we anticipate the following take of green (North Atlantic and South Atlantic DPSs), leatherback, and hawksbill sea turtles, Nassau grouper, and ESA-listed corals in the action area:

- 5,173 ESA-listed coral species of which two colonies may suffer lethal or non-lethal take annually from collisions with towed equipment or towed MEC/MPPEH, 50 may suffer lethal or non-lethal take from tissue sampling over 20 years, 4,656 may suffer non-lethal take from transplant stress, 517 may suffer lethal take from mortality due to transplant stress; and all colonies may suffer lethal or non-lethal take from underwater detonations, if BIPS or nonintentional detonations occur
- 10 juvenile green sea turtles, one juvenile hawksbill sea turtle, and two juvenile Nassau grouper may suffer non-lethal take from capture in cast nets or fish traps
- 103 green sea turtle hatchlings, 100 leatherback hatchlings, and 202 hawksbill hatchlings may be entrapped by in-water structures seaward of a nesting beach with five of the green, five of the leatherback, and 10 of the hawksbill sea turtle hatchlings suffering mortality from predation

The take listed above does not include take resulting from noise and potential physical effects from underwater detonations, including BIPs and nonintentional detonations for which adverse effects are expected to occur but have not yet been quantified. This take will be determined during a step-down consultation. We anticipate mother-calf pairs and/or juvenile sperm whales; adult and hatchling leatherback sea turtles; adult, juvenile, and hatchling green and hawksbill sea turtles; and adult and juvenile Nassau grouper will experience lethal or non-lethal take as a result of underwater detonations from BIPs or nonintentional detonation should these occur during removal activities in certain years over the 20-year lifetime of the action. Similarly, we anticipate take to elkhorn and staghorn critical habitat from underwater detonations, encapsulation, and removal of encrusted items. Depending on the extent of habitat impacts, there could be take of additional ESA-listed coral colonies or future recruitment. Any associated take would be part of future step-down consultations as well.

12.2 Reasonable and Prudent Measures

The RPMs described below are nondiscretionary, and must be undertaken by the Navy so that they become binding conditions for the exemption in section 7(o)(2) to apply. Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, RPMs and Terms and Conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified RPMs and Terms and Conditions identified in the Incidental Take Statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

Reasonable and prudent measures are nondiscretionary measures to minimize the amount or extent of incidental take (50 C.F.R. §402.02). NMFS believes the RPMs described below are necessary and appropriate to minimize the impacts of incidental take on green (North and South Atlantic DPS), hawksbill, and leatherback sea turtles, Nassau grouper, and ESA-listed corals:

1. An environmental monitoring plan shall be developed in coordination with NMFS and implemented prior to commencement of the installation of in-water structures seaward of nesting beaches that may present an entrapment hazard to sea turtle hatchlings or structures that may affect ESA-listed corals. These structures do not include any new boat access ramps, improvements to existing access ramps, or other in-water structures requiring the placement of fill, which would require step-down consultations.

2. Any marine lights installed and operated on in-water structures shall be in a sea turtle safe bandwidth to minimize potential hatchling disorientation.

3. Towing of MEC/MPPEH from underwater locations to terrestrial locations for disposal, as well as the operation of towed equipment shall be done in water depths and along

navigation routes selected to minimize potential collisions with ESA-listed coral colonies.

4. The locations of biological sampling using cast nets and fish traps shall be designed to minimize the potential bycatch of juvenile green and hawksbill sea turtles and Nassau grouper and the condition of released animals will be monitored. The use of nets other than cast nets will require step-down consultation.
5. Coral tissue sampling will preferentially be done on non-listed corals. If samples are collected from ESA-listed corals, methodology will be used to minimize the effects of tissue sampling on these colonies and the colonies will be opportunistically monitored to assess the effects of sampling.
6. The collection and transplant of ESA-listed corals prior to removal actions shall be undertaken in order to minimize the potential effects of removal activities on colonies growing on or immediately adjacent to suspected MEC/MPPEH. Transplanted colonies will be opportunistically monitored to assess their condition and transplant success.
7. The Navy must provide NMFS with all data collected during monitoring events and all monitoring reports.

12.3 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the Navy must comply with the following Terms and Conditions, which implement the RPMs described above. These include the take minimization, monitoring and reporting measures required by the section 7 regulations (50 C.F.R. §402.14(i)). These Terms and Conditions are non-discretionary. If the Navy fails to ensure compliance with these Terms and Conditions to implement the RPMs associated applicable to the authorities of the agency, the protective coverage of section 7(o)(2) may lapse. The terms and conditions detailed below for each of the RPMs include monitoring and minimization measures where needed.

1. The environmental monitoring plan for in-water structures will be designed in coordination with NMFS within 90 working days of the submittal of project-specific information to NMFS as described in this programmatic consultation. NMFS will have 30 working days to provide comments and recommendations on the plan such that the Navy can make any necessary edits prior to finalizing and implementing the plan for each proposed in-water structure. The plan will be implemented prior to any installation activities (RPM 1).
    a. The plan will include pre- and post-construction surveys to document the location of ESA-listed coral colonies relative to the final location of all in-water structures and their components such as anchors. The anchor point locations will be inspected prior to any installation activities to ensure no new ESA-listed coral

recruits are present. If recruits are present, the area will be surveyed to select a new anchor point or points.

b. All under- and above-water construction will be overseen by biologists who will have the authority to stop the work should effects to ESA-listed species described in this Opinion be observed that are different from or in excess of anticipated effects despite the implementation of the corresponding PDCs. Reinitiation may be required depending on the degree to which effects differ from those anticipated.

c. The plan will include a schedule for monitoring in-water structures, including monitoring following storm events to determine the stability of structures and their components such as anchors. Monitoring will also include an assessment of whether in-water structures lead to changes in navigation routes, shading, or gear interactions with ESA-listed coral recruits or coral critical habitat.

d. The plan will include contingency measures for the removal or redesign of structures if monitoring finds they are causing damage to ESA-listed coral colonies or their habitat due to changes in navigation routes associated with the presence of the structures. Reinitiation of consultation may be required if damage to ESA-listed coral colonies or their habitat from interactions with gear from the structures or shading of ESA-listed corals by the structure is observed because the effects of shading and gear interactions from structures was determined to be minimal based on previous consultations with the Navy for in-water structures in UXO 16.

e. The plan will also include monitoring for sea turtle nesting and hatchlings, as well as hatchling relocation should this be necessary.

    i. Beach monitoring will take place following USFWS recommendations before any installation activities should in-water structures be proposed seaward of nesting beaches. Beach monitoring will take place during the operational lifetime of the in-water structures as well. Should nesting occur, monitoring of the in-water structures will commence as soon as hatchlings begin to emerge and will continue daily for as long as hatchling emergence from nests in ongoing.

    ii. Monitoring of in-water structures for hatchling entrapment will include the removal of accumulated marine debris. Collected debris will be evaluated to ensure no hatchlings are mixed into the debris. All hatchlings trapped at an in-water structure, including those in accumulated marine debris, will be relocated to open sea beyond any structures.

    iii. The plan will include a schedule for monitoring activities and potential alternatives such as live-feed cameras in order to ensure rapid response to entrapment. The plan will also include contingency measures for the removal or redesign of the in-water structures should monitoring find that the structures are affecting the movement of hatchling sea turtles. Also, if in-water structures are found to affect other life stages of sea turtles not discussed in this Opinion, such as adults and juveniles through entanglement or entrapment, reinitiation of consultation may be required.

2. LEDs or other bulbs in a sea turtle safe bandwidth (greater than 560 nanometers) will be selected for use on any in-water structures requiring lights (RPM 2).
3. For MEC/MPPEH removal activities requiring that items be towed by a vessel to a terrestrial disposal site, and for surveys involving towed equipment, navigation routes shall be selected prior to commencement of work to the extent possible to minimize the potential for collisions with ESA-listed coral colonies, particularly those growing up from the seafloor or on hard substrate with higher relief that may be closer to the water surface (RPM 3).
    a. The route to be taken by the vessel towing the suspected MEC/MPPEH through any areas containing ESA-listed corals or elkhorn and staghorn coral critical habitat shall be selected in advance and provided as part of the project-specific data submission requirements detailed in this Opinion. Navigation routes will be selected that have adequate water depths under the vessel and item being towed, and expanses of seafloor without ESA-listed coral colonies in the swing radius of the tow rope and item to the maximum extent possible. The navigation routes will also have as few turns as possible in order to minimize slack in the line that could lead to items dropping lower in the water.
    b. The route to be taken by vessels towing survey or other equipment shall also be selected in advance and provided as part of the project-specific data submission requirements detailed in this Opinion. Contingency measures will be developed and implemented in case collisions occur despite implementation of the appropriate PDCs for underwater investigations.
    c. Any collisions with ESA-listed corals or coral habitats will be documented, including the location, water depth, vessel speed, weather and sea state, photographs and an assessment of the damage to ESA-listed coral colonies or elkhorn and staghorn coral critical habitat that includes the size of the impact area or measurements of the coral colony area damaged as a result of a collision. This information will be submitted to NMFS within 48 hours of any collisions.
4. The locations for placement of fish traps and throwing of cast nets will be carefully evaluated to minimize the potential for bycatch of juvenile sea turtles and Nassau grouper in these gear during biological sampling (RPM 4).

a. Because traps will be checked every 15 minutes and cast nets will be monitored continuously (Section 3.3.1), juvenile sea turtles and Nassau grouper caught in these gear should be released rapidly upon capture. Animals will be monitored following release to be sure they return to normal activities and do not appear to be in respiratory distress, be swimming erratically, or otherwise showing signs of distress following release. The plan for monitoring released animals will be developed in coordination with NMFS at least 90 working days prior to commencement of the first biological sampling event using cast nets and fish traps under this consultation. The plan will be implemented each time this sampling occurs.

b. In the case of sea turtles, if monitoring following release from fishing gear indicates animals are in distress, turtles will be recaptured and brought into the work vessel where they will be kept wet and the appropriate stranding network contacted to determine where to transport the animal for treatment.

c. If post-release monitoring indicates that captured sea turtles and Nassau grouper are suffering from stress or suffer mortality, reinitiation of consultation may be required. Modifications to the sampling design, methodology, and gear may also be made in order to minimize the capture and response to capture of these animals.

d. If a sea turtle is entangled in lines from fishing gear, divers will attempt to cut the gear and free the sea turtle underwater. If this is not possible, the turtle will gently be brought close to the vessel and a dip net or firm hold on the front flippers will be used to lift the turtle out of the water and onto the vessel where all gear will be gently cut from around the sea turtle to free the animal. Entangled turtles should never be lifted from the water by pulling on the gear in which they are entangled because this could result in injury. All entangled sea turtles, even if freed without incident, should be reported to the appropriate stranding network and wildlife agencies. If a sea turtle needs to be brought onboard a vessel either to disentangle it or to transport it for veterinary care, the turtle should be kept damp and in the shade and the local wildlife agency should be contacted immediately. Entanglement of all life stages of sea turtles or Nassau grouper in lines associated with fishing gear used during biological sampling may require reinitiation of consultation.

5. ESA-listed corals from which tissue samples are collected will be opportunistically monitored to assess their condition over time following sampling collection (RPM 5).

   a. A plan for opportunistically monitoring ESA-listed coral colonies from which tissue samples are collected will be developed in coordination with NMFS at least 90 working days prior to the first coral tissue sample collection under this consultation. Prior to sample collection, the location, species, and size, including diameter and branch length for branching corals and diameter for massive corals will be recorded. This information will be provided to NMFS as part of the annual reporting requirements under this programmatic consultation. Corals will be monitored opportunistically for up to two years following sampling collection when work is being performed by divers in waters where these corals are located. At a minimum, the plan will include measurements of the coral colonies, documentation of regrowth of the sampled area including photographs, and documentation of any disease or bleaching of the coral, including around the area from which a tissue sample was collected. The same measurements will be collected from colonies of the same species in the area to compare the condition of these colonies with sampled colonies.

   b. Should monitoring, particularly the comparison of sampled and non-sampled ESA-listed coral colonies, indicate that ESA-listed coral colonies from which samples were collected suffer from severe disease or bleaching around the area from which a sample was collected after more than one opportunistic monitoring event has been completed, tissue sampling of ESA-listed corals will cease.

6. The Navy will evaluate whether ESA-listed corals growing on items to be removed or in the footprint of removal activities can be transplanted and will opportunistically monitor these corals in comparison with ESA-listed corals that were not transplanted to assess transplant success (RPM 6).

   a. Surveys to determine the number, species, size, and condition of ESA-listed corals growing on items or in areas where removal activities are proposed and the approximate number of these that qualify for transplant will be completed prior to a removal action. This information will be provided to NMFS as part of the annual reporting requirements under this programmatic consultation. The collection and transplant of ESA-listed corals will be done in accordance with the PDCs for transplanting coral (Section 3.3.1).

   b. A subset of transplanted ESA-listed corals and a subset of ESA-listed corals that were not transplanted at the same site (whether corals were transplanted back to the site where the removal action occurred or an alternate site) will be opportunistically monitored. The plan for opportunistically monitoring transplanted corals and comparing the condition of these with the same species of ESA-listed corals that were not transplanted will be developed in coordination with NMFS at least 90 working days prior to the first transplant of ESA-listed coral colonies under this consultation.

   c. Should monitoring indicate that mortality rates, disease, bleaching, or other conditions are worse in transplanted corals, the transplant methods will be assessed to determine whether changes are required to improve transplant success.

7. The Navy must provide NMFS with all data collected during monitoring events required under these terms and conditions, as well as any monitoring reports generated over the lifetime of the project and following project completion, including as part of the annual programmatic review (RPM 7).

13 Conservation Recommendations

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

The following conservation recommendations are discretionary measures that NMFS believes are consistent with this obligation and therefore should be carried out by the Navy:

1. NMFS recommends that the use of anchor systems other than the concrete bulk anchors be explored such as three pyramid anchors (such as Dor-Mor™) or helical anchors be explored for areas containing seagrass beds in order to reduce the potential impacts to and loss of habitat for green sea turtle and Nassau grouper where oceanographic and sediment characteristics allow.
2. NMFS recommends that the Navy transplant seagrass that will be within the footprint of anchors and other components of in-water structures. A transplant and monitoring plan should be designed in coordination with NMFS, including the Habitat Conservation Division, for implementation prior to commencement of in-water structures.
3. NMFS recommends that uncolonized sand bottom areas be identified and the information marked on nautical charts and provided to contractors for anchoring of work vessels in project-specific work areas associated with all activities that are part of the proposed action.

In order for NMFS Office of Protected Resources Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, the Navy should notify the Interagency Cooperation Division of any conservation recommendations they implement in their final action.

14 Reinitiation Notice

This concludes formal programmatic consultation for the Navy for the investigation and the implementation of removal/remedial actions to address underwater munitions in UXO 16 around Vieques Island, Puerto Rico. Consistent with 50 C.F.R. §402.16(a), reinitiation of formal consultation is required and shall be requested by the Federal agency or by the Service, where discretionary Federal involvement or control over the action has been retained or is authorized by law and:

(1) The amount or extent of taking specified in the incidental take statement is exceeded.
(2) New information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not previously considered.
(3) The identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in this Opinion.
(4) A new species is listed or critical habitat designated under the ESA that may be affected by the action.

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