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

12/08/2020

F/SER31:HA
SERO-2020-00646

Chief, Miami Permits Section
Jacksonville District Corps of Engineers
Department of the Army
9900 Southwest 107th Avenue, Suite 203
Miami, Florida 33176

Ref.: SAJ-2002-05344 (LP-CGK), Sussman Dock, St. John, U.S. Virgin Islands

Dear Sir or Madam:

The enclosed Biological Opinion (Opinion) was prepared by the National Marine Fisheries Service (NMFS) pursuant to Section 7(a)(2) of the Endangered Species Act (ESA). The Opinion considers the effects of a proposal by the Jacksonville District of the United States Army Corps of Engineers (USACE) to authorize construction of a residential dock. NMFS concludes that the proposed action may affect, but is not likely to adversely affect, green sea turtle (North and South Atlantic distinct population segments [DPSs]), hawksbill sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), leatherback sea turtle, scalloped hammerhead shark (Central and Southwest Atlantic DPS), Nassau grouper, and giant manta ray. NMFS concludes that the proposed action is likely to adversely affect, but will not jeopardize the continued existence of elkhorn coral, staghorn coral, mountainous star coral, lobed star coral, and pillar coral. NMFS also concludes that the proposed action is likely to adversely affect, but will not destroy or adversely modify elkhorn and staghorn coral critical habitat.

The project has been assigned the tracking number SERO-2020-00646 in our new NMFS Environmental Consultation Organizer (ECO). Please refer to the ECO number in all future inquiries regarding this consultation. Please direct questions regarding this Opinion to Helena Antoun, Consultation Biologist, by phone at (939) 438-3123, or by email at Helena.antoun@noaa.gov.

Sincerely,

CRABTREE.ROY.E.DR.1365849559 Digitally signed by CRABTREE.ROY.E.DR.1365849559 Date: 2020.12.08 13:14:01 -05'00'

Roy E. Crabtree, Ph.D.
Regional Administrator

Enclosure:
Biological Opinion

File: 1514-22.F.10

**Endangered Species Act - Section 7 Consultation**
**Biological Opinion**

| | |
|---|---|
| **Action Agency**: | U.S. Army Corps of Engineers, Jacksonville District |
| **Applicant:** | Donald Sussman |
| | Permit Number SAJ-2002-05344 (LP-CGK) |
| **Activity**: | Dock construction, St. John, U.S. Virgin Islands |
| **Consulting Agency**: | National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida |
| | Consultation Tracking Number SERO-2020-00646 |
| **Approved By:** | CRABTREE.ROY.E.DR.1365849559 Digitally signed by CRABTREE.ROY.E.DR.1365849559 Date: 2020.12.08 13:15:22 -05'00'<br>Roy E. Crabtree, Ph.D., Regional Administrator<br>NMFS, Southeast Regional Office<br>St. Petersburg, Florida |
| **Date Issued**: | 12/08/2020 |

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**Acronyms and Abbreviations**

| | |
|---|---|
| CFR | Code of Federal Regulations |
| DPS | Distinct Population Segment |
| ECO | NMFS Environmental Consultation Organizer |
| ESA | Endangered Species Act |
| GHG | Green House Gas |
| ITS | Incidental Take Statement |
| MHW | Mean High Water |
| NMFS | National Marine Fisheries Service |
| NOAA | National Oceanic and Atmospheric Administration |
| NOS | National Ocean Service |
| Opinion | Biological Opinion |

| | |
|---|---|
| PCTS | Public Consultation Tracking System |
| PRD | NMFS Southeast Regional Office Protected Resources Division |
| RPA | Reasonable and Prudent Alternative |
| RPM | Reasonable and Prudent Measure |
| TNC | The Nature Conservancy |
| U.S. | United States |
| USACE | U.S. Army Corps of Engineers |
| USVI | U.S. Virgin Islands |

**Units of Measurement**

| | |
|---|---|
| ac | acre(s) |
| ft | foot/feet |
| $ft^2$ | square foot/feet |
| in | inch(es) |
| lin ft | linear foot/feet |
| m | meter(s) |
| mi | miles |
| $mi^2$ | square miles |
| km | kilometers |
| $km^2$ | kilometers square |
| $yd^3$ | cubic yard(s) |

## Introduction

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. § 1531 et seq.), requires that each federal agency ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. Section 7(a)(2) requires federal agencies to consult with the appropriate Secretary in carrying out these responsibilities. The National Oceanic and Atmospheric Administration (NOAA) National Marine Fisheries Service (NMFS) and the United States Fish and Wildlife Service share responsibilities for administering the ESA.

Consultation is required when a federal action agency determines that a proposed action "may affect" listed species or designated critical habitat. Informal consultation is concluded after NMFS determines that the action is not likely to adversely affect listed species or critical habitat. Formal consultation is concluded after NMFS issues a Biological Opinion ("Opinion") that identifies whether a proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify critical habitat, in which case reasonable and prudent alternatives to the action as proposed must be identified to avoid these outcomes. The Opinion states the amount or extent of incidental take of the listed species that may occur, develops measures (i.e., reasonable and prudent measures) to reduce the effect of take, and recommends conservation measures to further the recovery of the species. No destruction or adverse modification of critical habitat may be authorized, and thus there are no reasonable and prudent measures to minimize the destruction or adverse modification of critical habitat.

This document represents NMFS's Opinion based on our review of impacts associated with the proposed action within St. John, U.S. Virgin Islands (USVI). This Opinion analyzes the project's effects on threatened and endangered species and designated critical habitat, in accordance with Section 7 of the ESA. We based our Opinion on project information provided by the Jacksonville District of the U.S. Army Corps of Engineers (USACE) and other sources of information, including the published literature cited herein.

# 1 CONSULTATION HISTORY

The following is the consultation history for Environmental Consultation Organizer (ECO) tracking number SERO-2020-00646, Sussman Dock Construction:

- On March 19, 2020, NMFS received a request for formal consultation under section 7 of the ESA from USACE for construction permit application SAJ-2002-05344 (LP-CGK) in a letter dated March 19, 2020. Consultation was initiated that day.
- On September 16, 2020, NMFS received an email from USACE clarifying that only non-ESA-listed corals were to be relocated and transplanted from the action area of the proposed dock. The Nature Conservancy (TNC) will supply elkhorn, staghorn, mountainous star and lobed star corals from their nursery in St. Croix for out-planting; collection of corals of opportunity for this project will consist only of pillar corals.

# 2 DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA

## 2.1 Proposed Action

The USACE proposes to permit the applicant to install a new dock, which will extend 495 square feet ($ft^2$) (6 ft x 82.5 ft) waterward of the mean high water line (MHW), adjacent to a private estate. The USACE previously permitted a dock under the same permit number. The completed dock was not built to compliance and the USACE issued a tolling agreement, which included the removal of the dock. However, before the dock was removed, Hurricanes Maria and Irma in 2017 destroyed the dock leaving remnants that will be removed via barge as part of the proposed project.

The new dock will have molded composite grated decking and will be supported by 37 12-inch (in) diameter concrete piles. Of the 37 piles:

- 7 will be installed in sandy substrate,
- 6 will be installed in unconsolidated hardbottom,
- 19 will be installed in consolidated hardbottom, and
- 5 will be installed in un-colonized, intertidal hardbottom.

The proposed dock will be 4.2 feet (ft) above MHW. Construction will take place from a barge. The construction barge will be positioned in pre-established locations so as to not impact any coral colony or submerged aquatic vegetation (SAV). All piles will be installed via vibratory hammer in open water. In-water work is expected to take 5 to 7 days to complete during daylight hours only. The applicant will comply with NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions*[1] and will use turbidity curtains during construction. The proposed dock will have the potential to moor one vessel.

[1] NMFS. 2006. Sea Turtle and Smalltooth Sawfish Construction Conditions revised March 23, 2006. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, Saint Petersburg, Florida. http://sero.nmfs.noaa.gov/protected_resources/section_7/guidance_docs/documents/sea_turtle_and_smalltooth_sawfish_construction_conditions_3-23-06.pdf, accessed June 2, 2017.

The installation of the 19 piles in consolidated hardbottom (which is the essential feature of elkhorn and staghorn coral critical habitat) will damage approximately 15 $ft^2$ of substrate of suitable quality and availability for coral designated critical habitat[2]. As compensatory mitigation for that damage, the applicant proposes to out-plant up to 160 ESA-listed coral colonies comprised of elkhorn, staghorn, lobed star, and mountainous star corals obtained from TNC coral nursery located in St. Croix, USVI. In addition to these 160 ESA-listed coral colonies, the applicant's environmental consultant will collect up to 2 pillar coral fragments as corals of opportunity (naturally occurring unattached coral fragments) from the project site. The 2 pillar coral fragments will be housed at the TNC nursery and used to propagate up to 40 new pillar coral colonies. Thus, in total, the applicant proposes to out-plant up to 200 ESA-listed coral colonies to compensate for damaged coral critical habitat, which will be comprised of 40 pillar coral colonies, along with an undetermined combination of 160 ESA-listed coral colonies comprised of elkhorn, staghorn, lobed star, and mountainous star corals. The ESA-listed corals will be out-planted to the proposed 0.29-acre (ac) out-planting site to the west of the proposed dock (see Figure 1). Corals will be out-planted in habitat that is of the same depth, substrate and water quality as the proposed dock.


**Figure 1. Site of proposed dock (yellow diamond) and coral out-planting area (blue rectangle). Photo was provided by the action agency and taken prior to Hurricanes Irma and Maria.**

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[2] 19 piles x $(0.5\ ft)^2$ x 3.14 = 14.9 $ft^2$ (rounded up to 15 $ft^2$) of total area adversely affected by pile installation.

The proposed out-plant site currently supports various species of star coral, elkhorn and staghorn coral, as well as other non-ESA-listed corals. The proposed coral out-plants will be attached directly onto the available consolidated hardbottom with 2-part epoxy or cement. The proposed out-planting site is submerged land controlled by the USVI Department of Planning and Natural Resources (DPNR), and as a result, USVI DPNR will supervise the proposed out-planting of ESA-listed corals.

After out-planting is complete, divers will survey the coral out-planting site on a bi-weekly basis for the first 2 months after the transplant to ensure that the corals have not become unattached or shifted. If for any reason the corals become loose or move, they will be re-situated and/or reattached. After the first 2 months, the corals will be monitored on a monthly basis for the first year, making sure that they have remained stable and not shifted, and that corals have not come loose. If necessary, out-planted corals will be repositioned and re-attached.

As per the guidelines set forth in 40 CFR 230.96, the compensatory mitigation project will be monitored for a minimum of 5 years. A minimum of 75 monitoring quadrats will be established in the coral recipient/out-planting site, and at least 50 of the out-plants will be monitored. A baseline report will be prepared. The quadrats will be marked with a large cattle tag and photographed on a monthly basis for a period of 12 months after the out-planting. After the first year, the out-planted species will be monitored biannually for the remainder of the 5-year monitoring period. Reports will be provided with the photographs to the reviewing agencies within 30 days of the survey. A survey of all species (e.g., Nassau grouper) utilizing the area will be documented in the monitoring reports. The out-planting site is controlled and managed by the USVI Government; hence, any future alteration to the area would require both USACE and USVI Coastal Zone Management permits. Therefore, no future construction activities are anticipated in the action area.

### 2.2 Action Area

The proposed project site is located within the Great Cruz Bay at Parcels 300-40 and 300-50, St. John, USVI (18.318004°N, 64.791445°W [North American Datum 1983 (NAD83)]) (see Figure 1).

The action area is situated north of Blasbalg Point on the south side of Great Cruz Bay, adjacent to a private estate known as Chocolate Hole. Depth within the action area is from 0 ft (shoreline) to 15 ft. The substrate consists of rocky shoreline, sandy bottom, consolidated and unconsolidated hard bottom. A biological assessment was performed in September 2019. Coral colonized boulders and coral heads were identified scattered within the sand belt, dense submerged aquatic vegetation (SAV) composed of turtle grass, manatee grass, shoal grass and broadleaf grass were present. ESA-listed corals (specifically, 1 colony of elkhorn coral and 10 colonies of mountainous star coral) were identified and mapped within 15 ft of the proposed dock footprint. All 7 ESA-listed corals were identified in surveys east and west of the proposed project site. Eleven non-ESA listed corals were identified within footprint of the proposed dock. No mangroves were present.

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 Code of Federal Regulations [CFR] 402.02). As such, the action area includes the areas in which construction and coral out-planting will take place, as well as the immediate surrounding areas that may be affected by the proposed action. Based on our noise analysis below, the action area is equivalent to the radius of behavioral noise effects to ESA-listed fishes based on the proposed action's installation of 37 12-in concrete piles using a vibratory hammer (i.e., 705 ft behavioral noise radius; Figure 2).

**Figure 2. Image showing the action area defined by the extent of behavioral noise effects, red circle, based on the proposed action's installation of 12-inch concrete piles using a vibratory hammer. (ArcGIS)**

## 3 STATUS OF LISTED SPECIES AND DESIGNATED CRITICAL HABITAT

Table 1 provides the effect determinations for ESA-listed species the USACE and/or NMFS believe may be affected by the proposed action.

**Table 1. Effects Determinations for Species the Action Agency and/or NMFS Believe May Be Affected by the Proposed Action**

| Species | ESA Listing Status[3] | Action Agency Effect Determination | NMFS Effect Determination |
|---|---|---|---|
| **Sea Turtles** | | | |
| Green (North Atlantic [NA] distinct population segment [DPS]) | T | NLAA | NLAA |
| Green (South Atlantic [SA] DPS) | T | NLAA | NLAA |
| Kemp's Ridley | E | NLAA | NE |
| Leatherback | E | NLAA | NLAA |
| Loggerhead (Northwest Atlantic [NWA] DPS) | T | NLAA | NLAA |
| Hawksbill | E | NLAA | NLAA |
| **Fish** | | | |
| Scalloped hammerhead shark (Central and Southwest Atlantic DPS) | T | NLAA | NLAA |
| Oceanic whitetip shark | T | NLAA | NE |
| Nassau grouper | T | NLAA | NLAA |
| Giant manta ray | T | NLAA | NLAA |
| **Corals** | | | |
| Elkhorn coral | T | LAA | LAA |
| Staghorn coral | T | NE | LAA |
| Boulder star coral | T | LAA | NE |
| Mountainous star coral | T | LAA | LAA |
| Lobed star coral | T | NE | LAA |
| Pillar coral | T | LAA | LAA |
| **Mammals** | | | |
| Blue whale | E | NLAA | NE |
| Fin whale | E | NLAA | NE |
| Sei whale | E | NLAA | NE |

We believe the project will have no effect on Kemp's ridley sea turtles, oceanic whitetip shark, blue whale, fin whale and sei whale. We do not expect Kemp's ridley sea turtles to be present in the action area. Kemp's ridleys' distribution is throughout coastal waters of the Gulf of Mexico and the U.S. Atlantic Ocean from Florida to New England. This species is not known to occur in the USVI. Oceanic whitetip sharks are pelagic species; thus, we would not expect them to be present in the action area. Blue whales, fin whales, and sei whales are less common in the

[3] E = endangered; T = threatened; NLAA = may affect, not likely to adversely affect; NE = no effect; LAA = may affect, and is likely to adversely affect.

tropics and usually found in deep oceanic waters, which are not present in the action area. The Biological Assessment identified lobed star coral (*O. annularis*) as present within the action area; however, boulder star coral (*O. franksi*) was not identified as present within the action area where dock construction will take place. Thus, we believe the proposed action will have no effect on boulder star coral.

Table 2 provides the effects determinations for designated critical habitat occurring in the action area that the USACE and/or NMFS believe may be affected by the proposed action.

**Table 2. Effects Determinations for Designated Critical Habitat the Action Agency and/or NMFS Believe May Be Affected by the Proposed Action**

| Critical Habitat | Unit | USACE Effect Determination | NMFS Effect Determination |
|---|---|---|---|
| Elkhorn and staghorn coral | St. Thomas/St. John Unit | LAA | Likely to adversely affect, will not destroy or adversely modify |

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

Effects to ESA-listed sea turtles (green sea turtles [NA and SA DPSs], hawksbill sea turtles, loggerhead sea turtles [NWA DPS], and leatherback sea turtles), ESA-listed fish (scalloped hammerhead shark [Central and Southwest Atlantic DPS], Nassau grouper, and giant manta ray), and ESA-listed corals (elkhorn coral, staghorn coral, mountainous star coral, lobed star coral, and pillar coral) include the potential for injury from construction equipment or materials. We believe the risk of injury to ESA-listed sea turtles and ESA-listed fish is extremely unlikely to occur due to the species' ability to move away from the project site and into adjacent suitable habitat, if disturbed. In order to avoid impacts to ESA-listed corals, predetermined locations will be established for the construction barge to be set at locations without coral or submerged aquatic vegetation (SAV) colonization; hence, we believe the risk of injury to ESA-listed corals is extremely unlikely to occur. Limiting construction to daylight hours only will help construction workers regularly monitor for ESA-listed species near the project area and avoid interactions with these species. The applicants' implementation of NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* will further reduce the risk of injury with the requirement that all work be ceased if a sea turtle is observed within 50 ft from the operating or moving equipment.

The action area contains habitat that may be used by ESA-listed sea turtles and ESA-listed fish. ESA-listed sea turtles and ESA-listed fish may be affected by their inability to access the habitat within the action area due to their avoidance of construction activities and physical exclusion from the project area due to blockage by turbidity curtains. We believe habitat displacement effects to ESA-listed sea turtles and ESA-listed fish will be insignificant given the proposed action will be temporary and intermittent (i.e., in-water work will be between 5 to 7 days and construction will occur during daylight hours only) and will only occur within a small area adjacent to otherwise open water and usable habitat. In addition, because these species are mobile, we expect that they will move away from construction activities and forage in adjacent areas with similar habitat.

ESA-listed sea turtles, ESA-listed fish, and ESA-listed corals may be affected by the permanent loss of 15 $ft^2$ of habitat due to pile installation for the new dock.[4] We believe the permanent loss of 15 $ft^2$ of habitat will be insignificant to sea turtles, ESA-listed fish, and ESA-listed corals given the proposed project's small area of impact, and the abundance of similar surrounding habitat available to the species in the area.

The proposed action may result in an increase in vessel traffic in the area from the construction of a new dock with the potential to moor 1 vessel. ESA-listed sea turtles and fish species may be affected by being struck by the additional vessel using the dock, as it may increase the risk of collisions with these species. Based on a recent NMFS analysis,[5] it would take an introduction of at least 200 new vessels to an area to result in a take of 1 sea turtle in any single year. In addition, little information exists on vessel interactions with species with primarily demersal (i.e., bottom-dwelling) habits, such as Nassau grouper and scalloped hammerhead shark, because these species are rarely at risk from vessels at the surface. Moreover, in general, vessel strikes of elasmobranch species, which includes giant manta rays are extremely rare. Thus, the potential effects on ESA-listed sea turtles and fish resulting from increased vessel traffic associated with 1 new vessel are extremely unlikely to occur.

Effects to ESA-listed sea turtles and fish species as a result of noise created by construction activities can physically injure animals in the affected areas or change animal behavior in the affected areas. Injurious effects can occur in 2 ways. First, immediate adverse effects can occur to listed species if a single noise event exceeds the threshold for direct physical injury. Second, effects can result from prolonged exposure to noise levels that exceed the daily cumulative exposure threshold for the animals, and these can constitute adverse effects if animals are exposed to the noise levels for sufficient periods. Behavioral effects can be adverse if such effects interfere with animals migrating, feeding, resting, or reproducing, for example. Our evaluation of effects to listed species as a result of noise created by construction activities is based on recent NMFS analysis.[6],[7] The noise analysis in these consultations evaluates effects to ESA-listed fish and sea turtles identified by NMFS as potentially affected in the table above.[8]

Based on our noise calculations, installation of concrete piles by vibratory hammer will not result in any form of injurious noise effects to ESA-listed species. In SAJ-82 and JAXBO, the noise source level used for these analyses were based on the vibratory installation of a 13-in steel pipe pile and the vibratory installation of 36-in concrete piles, respectively. These reflect a conservative approach since the installation of a 13-in steel pipe pile would be considerably

[4] See Section 6, "Effects of the Action on Critical Habitat" for calculations.
[5] Barnette, M. 2018. Threats and Effects Analysis for Protected Resources on Vessel Traffic Associated with Dock and Marina Construction. NMFS Southeast Regional Office Protected Resources Division Memorandum. October 31, 2018.
[6] NMFS. Biological Opinion on Regional General Permit SAJ-82 (SAJ-2007-01590), Florida Keys, Monroe County, Florida. June 10, 2014.
[7] NMFS. USACE Jacksonville District's Programmatic Biological Opinion (JAXBO) (SER-2015-17616), November 20, 2017.
[8] While NMFS does not have information regarding noise effects specific to scalloped hammerhead shark (Central and Southwest Atlantic DPS) and giant manta rays, we believe that effects to scalloped hammerhead shark (Central and Southwest Atlantic DPS) and giant manta rays from pile driving noise would be very similar to effects on smalltooth sawfish (which are considered in SAJ-82 and JAXBO), because these species are elasmobranchs and lack swim bladders.

louder than a similarly-sized concrete pile. Likewise, the installation of a 36-in concrete pile would be considerably louder than the installation of a 12-in concrete pile. Installation of 13-in steel piles via vibratory hammer could result in behavioral effects at radii of up to 16 ft (5 m) for sea turtles and up to 72 ft (22 m) for ESA-listed fishes. Installation of 36-in concrete piles via vibratory hammer could result in behavioral effects at radii of up to 705 ft (215 m) for fish larger than 102 grams. Given the mobility of sea turtles and ESA-listed fish species, we expect them to move away from noise disturbances. Because there is similar habitat nearby, we believe this effect will be insignificant. If an individual chooses to remain within the behavioral response zone, it could be exposed to behavioral noise impacts during pile installation. Since installation will occur only during the day, these species will be able to resume normal activities during quiet periods between pile installations and at night. Moreover, the in-water work associated with this project will occur for only a short period of time, which is not expected to last more than 1 week. Therefore, installation of concrete piles by vibratory hammer will not result in any injurious noise effect, and we anticipate any behavioral effects will be insignificant.

Elkhorn and mountainous star corals may be impacted by shading from increased turbidity due to sediment resuspension and redeposition during the proposed pile-driving work. This route of effect will be mitigated through the use of turbidity barriers. Thus, we believe that any effects from turbidity would be extremely unlikely to occur.

Elkhorn and mountainous star corals could be affected by the construction barge anchor spuds during pile and dock installation. In order to avoid and minimize this potential route of effect, the barge will be positioned in pre-established locations where elkhorn and mountainous star coral are not present. Hence, we believe the risk of injury to ESA-listed corals would be extremely unlikely to occur.

## 3.2 Status of Species and Designated Critical Habitat Likely to be Adversely Affected

### 3.2.1 General Threats Faced by All Coral Species

Pillar coral, elkhorn coral, staghorn coral, mountainous star coral, lobed star coral and designated critical habitat for elkhorn and staghorn corals are likely to be adversely affected by the proposed action. In the summaries that follow, the status of the ESA-listed species and their designated critical habitats that occur within the proposed action area considered in this Opinion, are described. More detailed information on the status and trends of these listed resources 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 these NMFS websites:

- http://sero.nmfs.noaa.gov/protected_resources/index.html
- http://www.nmfs.noaa.gov/pr/species/esa/index.htm

Corals face numerous natural and man-made threats that shape their status and affect their ability to recover. Either 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. The main concerns regarding impacts of global climate change on coral reefs generally, and on listed corals in particular, are the magnitude and the rapid pace of change in greenhouse gas (GHG) concentrations (e.g., carbon dioxide [$CO_2$] 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). Ocean acidification affects a number of biological processes in corals, including secretion of their skeletons.

### 3.2.2 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.

### 3.2.3 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 impacts 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 greenhouse gases (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.

### 3.2.4 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 as a result 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.

### 3.2.5 Trophic Level 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 coralivores from predator control.

In the Caribbean, parrotfishes can graze at rates of more than 150,000 bites per square meter per day (Carpenter 1986), and thereby remove up to 90-100% of the daily primary production of algae. 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.

### 3.2.6 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 sub lethal 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.

### 3.2.7 Nutrient Enrichment

Elevated nutrient concentrations in seawater affect corals through 2 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.

## 3.3 Status of Pillar Coral

On September 10, 2014, NMFS listed pillar coral as threatened (79 FR 53852).

### 3.3.1 Species Description and Distribution

Pillar coral forms cylindrical columns on top of encrusting bases. Colonies are generally grey-brown in color and may reach approximately 10 ft (3 m) in height. Polyps' tentacles remain extended during the day, giving columns a furry appearance.

Pillar coral is present in the western Atlantic Ocean and throughout the greater Caribbean Sea, though is absent from the southwest Gulf of Mexico (Tunnell 1988). 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 (Florida Fish and Wildlife Conservation Commission 2013). Pillar coral inhabits most reef environments in water depths ranging from approximately 3-75 ft (1-25 m), but it is most common in water between approximately 15-45 ft (5-15 m) deep (Acosta and Acevedo 2006; Cairns 1982; Goreau and Wells 1967).

### 3.3.2 Life History Information

Average growth rates of 0.7-0.8 in (1.8-2.0 cm) per year in linear extension have been reported in the Florida Keys (Hudson and Goodwin 1997) compared to 0.3 in (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% of 185 colonies surveyed in Colombia), while the amount of partial mortality per colony is generally low (average of 3% of tissue area affected per colony).

Pillar coral is a gonochoric broadcast spawning[9] 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 there have been no reports of juvenile colonies 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 2008b) 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.

### 3.3.3 Status of Population Dynamics

Information on pillar coral status and populations dynamics is spotty throughout its range. Comprehensive and systematic census and monitoring has not been conducted outside of Florida. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Pillar coral is uncommon but conspicuous with scattered, isolated colonies. It is rarely found in aggregations. In coral surveys, it generally has a rare encounter rate, low percent cover, and low density.

Information on pillar coral is most extensive for Florida. In surveys conducted between 1999 and 2017, pillar coral was present at 0% to 13% of sites surveyed, and average density ranged

[9] Parents only contain one gamete (egg or sperm), which are released into the water column for fertilization by another parent's gamete.

from 0.0002 to 0.004 colonies per $m^2$ (NOAA, unpublished data). In 2014, there were 714 known colonies of pillar coral along the Florida reef tract from southeast Florida to the Dry Tortugas. In 2014, pillar coral colonies began to suffer from disease most likely associated with multiple years of warmer than normal temperatures. By April 2018, 75% of recorded colonies had suffered complete mortality (K. Neely and C. Lewis, unpublished data). The majority of these colonies were lost from the northern portion of the reef tract (Figure 3).

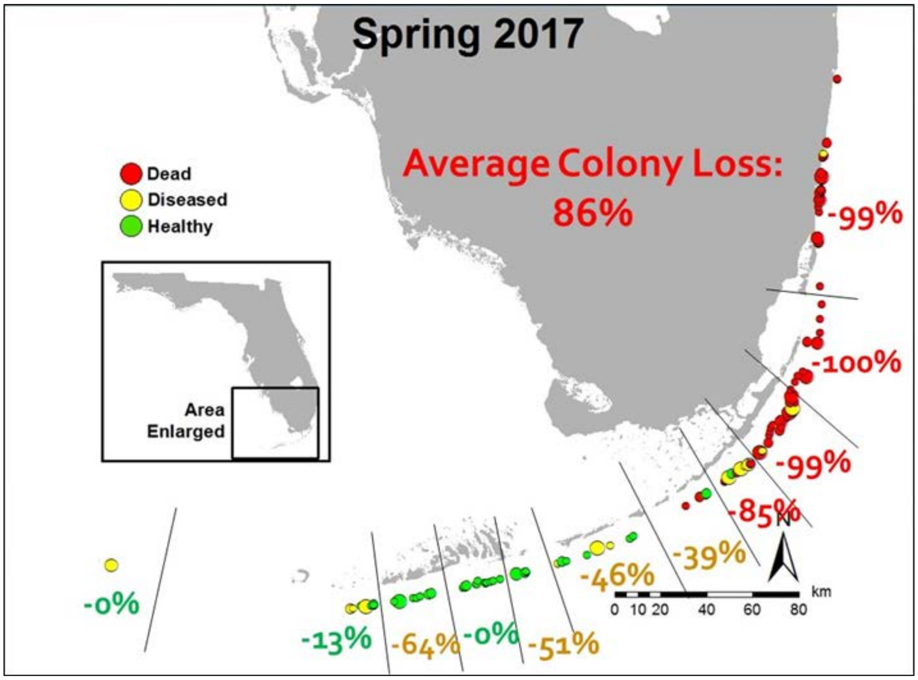


**Figure 3. Condition of known pillar coral colonies in Florida between 2014 and 2017 (Figure curtesy of K. Neely and C. Lewis).**

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 colonies per 10 $m^2$ (approximately 108 $ft^2$), 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 square miles) for an overall density of approximately 0.000017 colonies per 10 $m^2$ (approximately 100 $ft^2$) (Acosta and Acevedo 2006). In Puerto Rico, average density of pillar coral ranged from 0.0003 to 0.01 colonies per $m^2$ (approximately 100 $ft^2$); it occurred at 1% to 18% of the sites surveyed between 2008 and 2018 (NOAA unpublished data). In the U.S. Virgin Islands, average density of pillar coral ranged between 0.0003 and 0.005 colonies per $m^2$ (approximately 100 $ft^2$); it occurred in 1% to 6% of the sites surveyed between 2002 and 2017 (NOAA unpublished data). In Dominica, pillar coral comprised less than 0.9% cover and was present at 13% of 31 surveyed sites (Steiner 2003b). Pillar coral was observed on 1 of 7 fringing reefs surveyed off Barbados, and average cover was 3% (Tomascik and Sander 1987).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 46%

to 77% of pillar corals were impacted (NOAA 2018). In a post-hurricane survey of 57 sites in Florida, no pillar coral colonies were encountered, likely reflecting their much reduced population from disease (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands though qualitative observations indicate that damage was also widespread but variable by site.

Other than the declining population in Florida, there are 2 reports of population trends from the Caribbean. In monitored photo-stations in Roatan, Honduras, cover of pillar coral increased slightly from 1.35% in 1996 to 1.67% in 1999 and then declined to 0.44% in 2003 and to 0.43% in 2005 (Riegl et al. 2009). In the U.S. Virgin Islands, 7% of 26 monitored colonies experienced total colony mortality between 2005 and 2007, though the very low cover of pillar coral (0.04%) remained relatively stable during this time period (Smith et al. 2013).

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 studies that report pillar coral population trends indicate some decline with severe declines in Florida. Low density and gonochoric broadcast spawning reproductive mode, coupled with no observed sexual recruitment, indicate that natural recovery potential from mortality is low.

### 3.3.4 Threats

A summary of threats to all corals is provided in Section 3.2.1 General Threats Faced by All Coral Species. Detailed information on the specific threats to pillar coral can be found in the Final Listing Rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Pillar coral is susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and the trophic effects of fishing.

Pillar coral appears to have some susceptibility to ocean warming, though there are conflicting characterizations of the susceptibility of pillar coral to bleaching. Some locations experienced high bleaching of up to 100% of pillar coral colonies during the 2005 Caribbean bleaching event (Oxenford et al. 2008) while others had a smaller proportion of colonies bleach (e.g., 36%; Bruckner and Hill 2009). Reports of low mortality after less severe bleaching indicate potential resilience, though mortality information is absent from locations that reported high bleaching frequency. Although bleaching of most coral species is spatially and temporally variable, understanding the susceptibility of pillar coral is further confounded by the species' rarity and, hence, low sample size in any given survey.

Pillar coral is sensitive to cold temperatures. In laboratory studies of cold shock, pillar coral had the most severe bleaching of the 3 species tested at 12°C (Muscatine et al. 1991). During the 2010 cold water event in the Florida Keys, pillar coral experienced 100% mortality on surveyed inshore reefs, while other species experienced lower mortality (Kemp et al. 2011).

Pillar coral is susceptible to black band disease and white plague, though impacts from white plague are likely more extensive because of rapid progression rates (Brainard et al. 2011). Disease appears to be present in about 3-4% of pillar coral populations in locations surveyed

(Acosta and Acevedo 2006; Ward et al. 2006). Because few studies have tracked disease progression in pillar coral, the effects of disease are uncertain at both the colony and population level. However, in Florida where all known colonies of pillar coral were regularly monitored, extensive partial and whole colony mortality due to disease occurred in a large portion of the reef tract, reducing the overall number of pillar coral colonies in Florida by 57% and virtually eliminating pillar coral from the northern-most portion of its range (Figure 3).

Pillar coral appears to be moderately capable of removing sediment from its tissue (Brainard et al. 2011). However, pillar coral may be more sensitive to turbidity due to its high reliance on nutrition from photosynthesis (Brainard et al. 2011) and as evidenced by the geologic record (Hunter and Jones 1996). Pillar coral may also be susceptible to nutrient enrichment as evidenced by its absence from eutrophic sites in Barbados (Brainard et al. 2011), but there is uncertainty about whether its absence is a result of eutrophic conditions or a result of its naturally uncommon or rare occurrence. We anticipate that pillar coral likely has some susceptibility to sedimentation and nutrient enrichment. The available information does not support a more precise description of its susceptibility to this threat.

### 3.3.5 Summary of Status

Pillar coral is susceptible to a number of threats, and there is evidence of population declines in some locations and severe declines in Florida. 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. *Dendrogyra cylindrus* inhabits most reef environments in water depths ranging from 3-82 ft (1-25 m), but is naturally rare. 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. Also, low sexual recruitment, combined with its gonochoric, broadcast spawning reproduction mode and low density, 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.

## 3.4 Status of Elkhorn Coral

Elkhorn coral was listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing its status from threatened to endangered (77 FR 73219). On September 10, 2014, NMFS determined that elkhorn coral should remain listed as threatened (79 FR 53852).

### 3.4.1 Species Description and Distribution

Elkhorn coral colonies have frond-like branches, which appear flattened to near round, and typically radiate out from a central trunk and angle upward. Branches are up to approximately 20 in (50 cm) wide and range in thickness from about 1.5-2 in (4 to 5 cm). Individual colonies can grow to at least 6.5 ft (2 m) in height and 13 ft (4 m) in diameter (*Acropora* Biological Review Team 2005). Colonies of elkhorn coral can grow in nearly single-species, dense stands and form an interlocking framework known as thickets.

Elkhorn coral is distributed throughout the western Atlantic Ocean, Caribbean Sea, and Gulf of Mexico. The northern extent of the range in the Atlantic is Broward County, Florida, where it is relatively rare (only a few known colonies), but fossil elkhorn coral reef framework extends into Palm Beach County, Florida. There are 2 known colonies of elkhorn coral, which were discovered in 2003 and 2005, at the Flower Garden Banks, which is located 100 miles (161 km) off the coast of Texas in the Gulf of Mexico (Zimmer et al. 2006). The species has been affected by extirpation from many localized areas throughout its range (Jackson et al. 2014).

Goreau (1959) described 10 habitat zones on a Jamaican fringing reef from inshore to the deep slope, finding elkhorn coral in 8 of the 10 zones. Elkhorn coral commonly grows in turbulent water on the fore-reef, reef crest, and shallow spur-and-groove zone (Cairns 1982; Miller et al. 2008; Rogers et al. 1982; Shinn 1963) in water ranging from approximately 3-15 ft (1-5 m) depth, and up to 40 ft (12m). Elkhorn coral often grows in thickets in fringing and barrier reefs (Jaap 1984; Tomascik and Sander 1987; Wheaton and Jaap 1988). They have formed extensive barrier-reef structures in Belize (Cairns 1982), the greater and lesser Corn Islands, Nicaragua (Lighty et al. 1982), and Roatan, Honduras, and extensive fringing reef structures throughout much of the Caribbean (Adey 1978). Early studies termed the reef crest and adjacent seaward areas from the surface down to approximately 20 ft (5-6 m) depth the "palmata zone" because of the domination by the species (Goreau 1959; Shinn 1963). It also occasionally occurs in back-reef environments and in depths up to 98 ft (30 m).

### 3.4.2 Life History Information

Relative to other corals, elkhorn coral has a high growth rate allowing 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 2-4 in (4-11 cm) per year (*Acropora* Biological Review Team 2005). Annual growth has been found to be dependent on the size of the colony, and new recruits and juveniles typically grow at slower rates. Additionally, stressed colonies and fragments may also exhibit slower growth.

Elkhorn coral is a hermaphroditic broadcast spawning[10] species that 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 $cm^2$), 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 self-fertilization is possible, elkhorn coral is largely self-incompatible (Baums et al. 2005a; 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 9 months are high based on settlement experiments (Szmant and Miller 2006). Laboratory studies have found that certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010). Laboratory experiments have shown that some individuals (i.e., genotypes) are sexually incompatible (Baums et al. 2013) and that the proportion of eggs

[10] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

fertilized increases with higher sperm concentration (Fogarty et al. 2012). Experiments using gametes collected in Florida and Belize showed that Florida corals had lower fertilization rates than those from Belize, possibly due to genotype incompatibilities (Fogarty et al. 2012).

Reproduction occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Bak and Criens 1982; Highsmith 1982; Lirman 2000; Miller et al. 2007; Wallace 1985). Storms can be a method of producing fragments to establish new colonies (Fong and Lirman 1995). Fragmentation is an important mode of reproduction in many reef-building corals, especially for branching species such as elkhorn coral (Highsmith 1982; Lirman 2000; Wallace 1985). However, in the Florida Keys where populations have declined, there have been reports of failure of asexual recruitment due to high fragment mortality after storms (Porter et al. 2012; Williams and Miller 2010; Williams et al. 2008).

The combination of relatively rapid skeletal growth rates and frequent asexual reproduction by fragmentation can enable effective competition within, and domination of, elkhorn coral in reef-high-energy environments such as reef crests. Rapid skeletal growth rates and frequent asexual reproduction by fragmentation facilitate potential recovery from disturbances when environmental conditions permit (Highsmith 1982; Lirman 2000). However, low sexual reproduction can lead to reduced genetic diversity and limits the capacity to repopulate sites distant from the parent.

### 3.4.3 Status of Population Dynamics

Information on elkhorn coral status and populations dynamics is spotty 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 were data exist.

There appear to be two distinct populations of 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. 2005b). 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. 2005b). Models suggest that the Mona Passage between the Dominican Republic and Puerto Rico acts as a filter for larval dispersal and gene flow between the eastern Caribbean and western Caribbean (Baums et al. 2006b).

The western Caribbean is characterized by genetically poor populations with lower densities ($0.13 \pm 0.08$ colonies per $m^2$). The eastern Caribbean populations are characterized by denser ($0.30 \pm 0.21$ colonies per $m^2$), genotypically richer stands (Baums et al. 2006a). Baums et al. (2006a) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment. They postulated these 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.  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. 2005b).  In contrast, all 15 colonies sampled in Navassa had unique genotypes (Baums et al. 2006a).  Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotpyes 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. 2006a).  In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24% and 65% of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006a).

A genetic study found significant population structure in Puerto Rico locations (Mona Island, Desecheo Island, La Parguerain, La Parguera) both between reefs and between locations.  The study suggests that there is a restriction of gene flow between some reefs in close proximity in the La Parguera reefs resulting in greater population structure (Garcia Reyes and Schizas 2010).  A more recent study provided additional detail on the genetic structure of elkhorn coral in Puerto Rico, as compared to Curaçao, the Bahamas, and Guadeloupe that found unique genotypes in 75% of the samples with high genetic diversity (Mège et al. 2014).  The recent results support two separate populations of elkhorn coral in the eastern Caribbean and western Caribbean; however, there is less evidence for separation at Mona Passage, as found by Baums et al. (2006b).

Elkhorn coral was historically one of the dominant species on Caribbean reefs, forming large, monotypic thickets and giving rise to the "elkhorn" zone in classical descriptions of Caribbean reef morphology (Goreau 1959).  However, mass mortality, apparently 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).  This mass mortality occurred throughout the range of the species within all Caribbean countries and archipelagos, even on reefs and banks far from localized human influence (Aronson and Precht 2001; Wilkinson 2008).  In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events added to the decline of elkhorn coral (Brainard et al. 2011).  In locations where historic quantitative data are available (Florida, Jamaica, U.S. Virgin Islands), there was a reduction of greater than 97% between the 1970s and early 2000s in elkhorn coral populations (Acropora Biological Review Team 2005).

Since the 2006 listing of elkhorn coral, continued population declines have occurred in some locations with certain populations of elkhorn coral decreasing up to an additional 50% or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008).  In addition, Williams et al. (2008) reported asexual recruitment failure between 2004 and 2007 in the upper Florida Keys after a major hurricane season in 2005 where less than 5% of the fragments produced recruited into the population.  In contrast, several studies describe elkhorn coral populations that are showing some signs of recovery or are stable including in the Turks and Caicos Islands (Schelten et al. 2006), U.S. Virgin Islands (Grober-Dunsmore et al. 2006; Mayor et al. 2006; Rogers and Muller 2012), Venezuela (Zubillaga et al. 2008), and Belize (Macintyre and Toscano 2007).

There is some density data available for elkhorn corals in Florida, Puerto Rico, the U.S. Virgin Islands, and Cuba. In Florida, elkhorn coral was detected at 0% to 78% 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 U.S. Virgin Islands from 2002 to 2017. It was observed at 0 to 7% 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, U.S. Virgin Islands 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 1% to 27% 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).

Mayor et al. (2006) reported the abundance of elkhorn coral in Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands. They surveyed 617 sites from May to June 2004 and extrapolated density observed per habitat type to total available habitat. Within an area of 795 ha, they estimated 97,232–134,371 (95% confidence limits) elkhorn coral colonies with any dimension of connected live tissue greater than one meter. Mean densities (colonies $\geq$ 1 m) were 0.019 colonies per $m^2$ in branching coral-dominated habitats and 0.013 colonies per $m^2$ in other hard bottom habitats.

Puerto Rico contains the greatest known extent of elkhorn coral in the U.S. Caribbean; however, the species is still rarely encountered. Between 2006 and 2007, a survey of 431 random points in habitat suitable for elkhorn coral in 6 marine protected areas in Puerto Rico revealed a variable density of 0-52 elkhorn coral colonies per 100 $m^2$, with average density of 0.03 colonies per $m^2$. Live elkhorn coral colonies were present at 31% of all points sampled, and total loss of elkhorn coral was evidenced in 14% of the random survey areas where only dead standing colonies were present (Schärer et al. 2009).

In stratified random surveys along the south, southeast, southwest, and west coasts of Puerto Rico designed to locate *Acropora* colonies, elkhorn coral was observed at 5 out of 301 stations with sightings outside of the survey area at an additional 2 stations (García Sais et al. 2013). Elkhorn coral colonies were absent from survey sites along the southeast coast. Maximum density was 18 colonies per 15 $m^2$ (1.2 colonies per $m^2$), and maximum colony size was approximately 7.5 ft (2.3 m) in diameter (García Sais et al. 2013).

Demographic monitoring of elkhorn coral colonies in Florida has shown a decline over time. Upper Florida Keys colonies showed more than 50% loss of tissue as well as a decline in the number of colonies, and a decline in the dominance by large colonies between 2004 and 2010 (Vardi et al. 2012; Williams and Miller 2012). Elasticity analysis from a population model based on data from the Florida Keys has shown that the largest individuals have the greatest contribution to the rate of change in population size (Vardi et al. 2012). Between 2010 and 2013, elkhorn coral in the middle and lower Florida Keys had mixed trends. Population densities remained relatively stable at 2 sites and decreased at 2 sites by 21% and 28% (Lunz 2013). Following the 2014 and 2015 thermal stress events, monitored elkhorn coral colonies lost one-third of their live tissue (Williams et al. 2017).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands 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% of elkhorn corals were impacted (NOAA 2018). Survey data for impacts to elkhorn corals are not available for the U.S. Virgin Islands or Florida, though qualitative observations indicate that damage was also widespread but variable by site.

At 8 of 11 sites in St. John, U.S. Virgin Islands, colonies of elkhorn coral increased in abundance, between 2001 and 2003, particularly in the smallest size class, with the number of colonies in the largest size class decreasing (Grober-Dunsmore et al. 2006). Colonies of elkhorn coral monitored monthly between 2003 and 2009 in Haulover Bay on St. John, U.S. Virgin Islands suffered bleaching and mortality from disease but showed an increase in abundance and size at the end of the monitoring period (Rogers and Muller 2012). The overall density of elkhorn coral colonies around St. John did not significantly differ between 2004 and 2010 with 6 out of the 10 sites showing an increase in colony density. Size frequency distribution did not significantly change at 7 of the 10 sites, with 2 sites showing an increased abundance of large-sized (> 51 cm) colonies (Muller et al. 2014).

In Curaçao, elkhorn coral monitored between 2009 and 2011 decreased in abundance and increased in colony size, with stable tissue abundance following hurricane damage (Bright et al. 2013). The authors explained that the apparently conflicting trends of increasing colony size but similar tissue abundance likely resulted from the loss of small-sized colonies that skewed the distribution to larger size classes, rather than colony growth.

Simulation models using data from matrix models of elkhorn coral colonies from specific sites in Curaçao (2006-2011), the Florida Keys (2004-2011), Jamaica (2007-2010), Navassa (2006 and 2009), Puerto Rico (2007 and 2010), and the British Virgin Islands (2006 and 2007) indicate that most of these studied populations will continue to decline in size and extent by 2100 if environmental conditions remain unchanged (i.e., disturbance events such as hurrricanes do not increase; Vardi 2011). In contrast, the studied populations in Jamaica were projected to increase in abundance, and studied populations in Navassa were projected to remain stable. Studied populations in the British Virgin Islands were predicted to decrease slightly from their initial very low levels. Studied populations in Florida, Curaçao, and Puerto Rico were predicted to decline to zero by 2100. Because the study period did not include physical damage (storms), the population simulations in Jamaica, Navassa, and the British Virgin Islands may have contributed to the differing projected trends at sites in these locations.

A report on the status and trends of Caribbean corals over the last century indicates that cover of elkhorn coral has remained relatively stable at approximately 1% throughout the region since the large mortality events of the 1970s and 1980s. The report also indicates that the number of reefs with elkhorn coral present steadily declined from the 1980s to 2000-2004, then remained stable between 2000-2004 and 2005-2011. Elkhorn coral was present at about 20% of reefs surveyed in both the 5-year period of 2000-2004 and the 7-year period of 2005-2011. Elkhorn coral was dominant on approximately 5 to 10% of hundreds of reef sites surveyed throughout the Caribbean during the 4 periods of 1990-1994, 1995-1999, 2000-2004, and 2005-2011 (Jackson et al. 2014).

Overall, frequency of occurrence decreased from the 1980s to 2000, stabilizing in the first decade of 2000. 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 numbers are decreasing. In some cases, when size class distribution is not reported, there is uncertainty of whether increases in abundance indicate growing populations or fragmentation of larger size classes into more small-sized colonies. From locations where size class distribution is reported, there is evidence of recruitment, but not the proportions of sexual versus asexual recruits. Events like hurricanes continue to heavily impact local populations and affect projections of persistence at local scales. We conclude there has been a significant decline of elkhorn coral throughout its range as evidenced by the decreased frequency of occurrence and that population abundance is likely to decrease in the future with increasing threats.

#### 3.4.4 Threats

Detailed information on the threats to elkhorn coral can be found in the Final Listing rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Elkhorn coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing, depensatory population effects from rapid, drastic declines and low sexual recruitment, and anthropogenic and natural abrasion and breakage.

Elkhorn coral is highly susceptible to disease as evidenced by the mass-mortality event in the 1970s and 1980s. White pox seems to be more common today than white band disease. The effects of disease are spatially and temporally (both seasonally and inter-annually) variable. Results from longer-term monitoring studies in the U.S. Virgin Islands and the Florida Keys indicate that disease can be a major cause of both partial and total colony mortality.

Elkhorn coral is highly susceptible to ocean warming. High water temperatures affect elkhorn coral through bleaching, lowered resistance to disease, and effects on reproduction. Temperature-induced bleaching and mortality following bleaching are temporally and spatially variable. Bleaching associated with the high temperatures in 2005 had a large impact on elkhorn coral with 40 to 50 % of bleached colonies suffering either partial or complete mortality in several locations. Algal symbionts did not shift in elkhorn coral after the 1998 bleaching event indicating the ability to adapt to rising temperatures may not occur through this mechanism. However, elkhorn coral showed evidence of resistance to bleaching from warmer temperatures in some portions of its range under some circumstances (Little Cayman). Through the effects on reproduction, high temperatures can potentially decrease larval supply and settlement success, decrease average larval dispersal distances, and cause earlier larval settlement affecting gene flow among populations.

Elkhorn coral is susceptible to acidification through reduced growth, calcification, and skeletal density. The effects of increased carbon dioxide combined with increased nutrients appear to be much worse than either stressor alone.

There are few studies of the effects of nutrients on elkhorn coral. Field experiments indicate that the mean net rate of uptake of nitrate by elkhorn coral exceeds that of ammonium by a factor of two and that elkhorn coral does not uptake nitrite (Bythell 1990). In Vega Baja, Puerto Rico, elkhorn coral mortality increased to 52% concurrent with pollution and sedimentation associated with raw sewage and beach nourishment, respectively, between December 2008 and June 2009 (Hernandez-Delgado et al. 2011a). Mortality presented as patchy necrosis-like and white pox-like conditions that impacted local reefs following anthropogenic disturbances and was higher inside the shallow platform (52-69%) and closer to the source of pollution (81-97%) compared to the outer reef (34 to 37 percent; Hernandez-Delgado et al. 2011a). Elkhorn coral is sensitive to nutrients as evidenced by increased mortality after exposure to raw sewage. Elkhorn coral is highly susceptible to nutrient enrichment. Elkhorn coral is also sensitive to sedimentation due to its poor capability of removing sediment and its high reliance on clear water for nutrition. Sedimentation can also cause tissue mortality.

Predators can have an impact on elkhorn coral both through tissue removal and the potential to spread disease. Predation pressure is spatially variable and almost non-existent in some locations. However, the effects of predation can become more severe if colonies decrease in abundance and density, as predators focus on the remaining living colonies.

### 3.4.5 Summary of Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events. Elkhorn 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 elkhorn coral is limited to an area with high localized human impacts and predicted increasing threats. Elkhorn coral occurs in turbulent water on the back reef, fore reef, reef crest, and spur and groove zone in water ranging from 1 to 30 m in depth. This moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that will, on local and regional scales, experience highly variable thermal regimes and ocean chemistry at any given point in time. Elkhorn 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 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 anticipate that the population abundance is likely to decrease in the future with increasing threats.

## 3.5 Status of Staghorn Coral

Staghorn coral was listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing its status from threatened to endangered (77 FR 73219). On September 10, 2014, NMFS determined that staghorn coral should remain listed as threatened (79 FR 53852).

### 3.5.1 Species Description and Distribution

Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges from 0.1-2 in (0.25-5 cm; Lirman et al. 2010), and linear branch growth rates have been reported to range between 1.2-4.5 in (3-11.5 cm) per year (*Acropora* Biological Review Team 2005). The species can exist as isolated branches, individual colonies up to about 5 ft (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 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 16 to 65 ft (5 to 20 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. 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 16-65 ft (5-20 m) in depth; though it has rarely been found to approximately 195 ft (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 (~53-99 ft [16-30 m]; Goldberg 1973). Historically, staghorn coral was one of the primary constructors of mid-depth (approximately 33-50 ft [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., 16-65 ft; 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 only 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. 2014).

### 3.5.2 Life History Information

Relative to other corals, staghorn coral has a high growth rate that have 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 2-4 in (4-11 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[11]. 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 6 in (17 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 the presence of certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010).

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.

### 3.5.3 Status of Population Dynamics

Information on staghorn 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.

Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from 9 regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 310 miles (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 1.25 miles (2 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 U.S. Virgin Islands, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the U.S. Virgin Islands and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

[11] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

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, U.S. Virgin Islands, Belize), there was a reduction of approximately 92 to greater than 97% 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 decreasing up to an additional 50% or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008). Some small pockets of remnant robust populations have been reported 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.

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% in 1996 to 0.14% 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% in 2005. Staghorn coral colony frequency decreased 71% between 1997 and 1999. In sharp contrast, offshore bank reefs near Roatan had dense thickets of staghorn coral with 31% 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).

Other studies of population dynamics show mixed trends. While cover of staghorn coral increased from 0.6% in 1995 to 10.5% in 2004 (Idjadi et al. 2006) and 44% in 2005 on a Jamaican reef, it collapsed after the 2005 bleaching event and subsequent disease to less than 0.5% 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%) (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 one of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50%, 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 the percentage of reefs with staghorn coral present has decreased over time. 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% of reefs and remained relatively stable at 30% through the 1990s. The number of reefs with staghorn coral present decreased to approximately 20% in 2000-2004 and approximately 10% in 2005-2011 (Jackson et al. 2014).

There is some density data available for reefs in U.S. jurisdiction. In Florida, staghorn coral was detected at 3% to 15% of the sites surveyed between 1999 and 2017. Average density ranged from 0.001 to 0.17 colonies per $m^2$. Staghorn coral was encountered less frequently during benthic surveys in the U.S. Virgin Islands from 2002 to 2017. It was typically observed at < 3% of surveyed reefs with the highest frequency of observance at 18% in 2012. Density ranged from <0.001 to 0.07 colonies per $m^2$ (NOAA, unpublished data).

Benthic surveys between 2008 and 2018 in Puerto Rico detected an average density of 0.001 to 0.17 colonies per $m^2$, and colonies were observed at 4% to 25% of the reefs surveyed (NOAA, unpublished data). 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 24 in (60 cm) and density ranged from 1-10 colonies per 162 $ft^2$ (15 $m^2$; García Sais et al. 2013).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 38% to 54% of staghorn corals were impacted (NOAA 2018). In a post-hurricane survey of 57 sites in Florida, all of the staghorn coral colonies encountered were damaged by the hurricane (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands though qualitative observations indicate that damage was also widespread but variable by site.

Overall, 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, 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 time frames. 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. We anticipate that population abundance is likely to decrease in the future with increasing threats.

### 3.5.4 Threats

Detailed information on the threats to staghorn coral can be found in the Final Listing rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Staghorn coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, as well as susceptible to trophic effects of fishing, depensatory population effects from rapid, drastic declines and low sexual recruitment, and anthropogenic and natural abrasion and breakage.

Staghorn coral is highly susceptible to disease as evidenced by the mass-mortality event in the 1970s and 1980s. Although disease is both spatially and temporally variable, about 5-6% of staghorn coral colonies appear to be affected by disease at any one time, though incidence of disease has been reported to range from 0-32% and up to 72% during an outbreak. There is indication that some colonies may be resistant to white band disease. Staghorn coral is also susceptible to several other diseases including one that causes rapid tissue loss from multiple lesions (e.g., Rapid Wasting Disease, White Patch Disease). Because few studies track diseased colonies over time, determining the present-day colony and population level effects of disease is difficult. One study that monitored individual colonies during an outbreak found that disease can be a major cause of both partial and total colony mortality (Williams and Miller 2005).

Staghorn coral is highly susceptible to bleaching in comparison to other coral species, and mortality after bleaching events is variable. Algal symbionts did not shift in staghorn coral after the 1998 bleaching event, indicating the ability of this species to acclimatize to rising temperatures may not occur through this mechanism. Data from Puerto Rico and Jamaica following the 2005 Caribbean bleaching event indicate that temperature anomalies can have a large impact on total and partial mortality and reproductive output.

Staghorn coral is highly susceptible to acidification through reduced growth, calcification, and skeletal density. The effects of increased carbon dioxide combined with increased nutrients appear to be synergistically worse and caused 100% mortality in some combination in one laboratory study.

Staghorn coral has high susceptibility to sedimentation through its sensitivity to turbidity (reduced light results in lower photosynthesis by symbiotic algae, so there is less food for the coral), and increased run-off from land clearing has resulted in mortality of this species through smothering. In addition, laboratory studies indicate the combination of sedimentation and nutrient enrichment appears to be synergistically worse.

Staghorn coral is also highly susceptible to elevated nutrients, which can cause decreased growth in staghorn coral. The combined effects of nutrients with other stressors such as elevated carbon dioxide and sedimentation appear to be worse than the effects of nutrients alone, and can cause colony mortality in some combinations.

Predators can have a negative impact on staghorn coral through both tissue removal and the spread of disease. Predation pressure appears spatially variable. Removal of tissue from growing branch tips of staghorn coral may negatively affect colony growth, but the impact is unknown as most studies do not report on the same colonies through time, inhibiting evaluation of the longer-term impact of these predators on individual colonies and populations.

### 3.5.5 Summary of Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. 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 5 to 20 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. 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.

## 3.6 Status of Mountainous Star Coral

On September 10, 2014, NMFS listed mountainous star coral as threatened (79 FR 53852). Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the 3 species in the *Orbicella annularis* (star coral) complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 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 highly plastic, single species –*Montastraea annularis*– with growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (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*. These 3 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 3 species.

Some studies report on the species complex rather than individual species because 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. Species-specific information is reported when available. 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 3 separate species.

### 3.6.1 Species Description and Distribution

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 2 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.

### 3.6.2 Life History Information

The star coral species complex has growth rates ranging from 0.02-0.5 in (0.06-1.2 cm) per year and averaging approximately 0.3 in (1 cm) linear growth per year. Mountainous star coral's growth rate is intermediate between the other star coral complex species (Szmant et al. 1997). They grow more slowly in deeper water and in water that is less clear.

The star coral complex species are hermaphroditic broadcast spawners,[12] as spawning is concentrated on 6-8 nights following the full moon in late August, September, or early October, depending on location and timing of full moon. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about 1-2 hours earlier. Fertilization success measured in the field was generally below 15% for all 3 species, as it is closely linked to the number of colonies concurrently spawning. In Puerto Rico, minimum size at reproduction for the star coral species complex was 12 $in^2$ (83 $cm^2$).

Successful recruitment by the star coral species complex has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 130 $ft^2$ (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.

[12] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

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% 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% remaining alive after 30 days. Post-settlement survivorship was much lower than the 29% observed for elkhorn coral after 7 months (Szmant and Miller 2005).

Mountainous star coral has slow growth rates, late reproductive maturity, and low recruitment rates. 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, we conclude that the buffering capacity of this life history strategy has been reduced by recent population declines and partial mortality, particularly in large colonies.

### 3.6.3 Status of Population Dynamics

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 3 genotypes (Levitan et al. 2011) potentially indicating 30% clonality.

Benthic surveys along the Florida Reef Tract between 1999 and 2017 have shown a decrease of mountainous star coral (NOAA, unpublished data). In 1999, mountainous star coral was present at 62% of surveyed sites and had an average density of 0.62 colonies per $m^2$. Presence and density decreased substantially after 2005, and in 2017, mountainous star coral was present at 30% of sites and had an average density of 0.09 colonies per $m^2$.

Benthic survey data for the U.S. Caribbean show less variability in the density of mountainous star coral. In Puerto Rico, average density was between 0.1 and 0.2 colonies per $m^2$ between 2008 and 2016 (NOAA, unpublished data). In 2018, average density was recorded as 0.01 colonies per $m^2$, the lowest recorded for all survey years. In the U.S. Virgin Islands, density ranged from 0.01 to 0.2 colonies per $m^2$ between 2002 and 2017 with no obvious trends among years.

Recent events have greatly impacted coral populations in Florida and the U.S. 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. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species, including mountainous star coral. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 12-14% of mountainous star corals were impacted (NOAA 2018). In Florida, approximately 24% of mountainous star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands though qualitative observations indicate that damage was also widespread but variable by site.

In the Flower Garden Banks, limited benthic surveys show density of mountainous star coral remained relatively stable between 2010 and 2015 (NOAA, unpublished data). Average density was recorded as 0.09 colonies per $m^2$ in 2010, 0.19 colonies per $m^2$ in 2013, and 0.21 colonies per $m^2$ in 2015. These may represent an increasing trend as the presence of mountainous star coral also increased during this same period. It was present at 35% of sites in 2010 and increased to 68% of sites in 2013 and 77% of sites in 2015.

Limited data are available for other areas of the Caribbean. On remote reefs off southwest Cuba, average density of mountainous star coral was 0.12 colonies per 108 $ft^2$ (10 $m^2$) at 38 reef-crest sites and 1.26 colonies per 108 $ft^2$ (10 $m^2$) at 30 reef-front sites (Alcolado et al. 2010). In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80% of the sites at 1-10% cover (Steiner 2003a).

Population trend data exists for several locations. At 9 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% and 48% at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27% of all corals at 6 sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11% in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12% of all coral colonies were mountainous star coral in 2000, compared to 7% in 2008.

In a survey of 185 sites in 5 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%. The total live area occupied by mountainous star coral declined by a mean of 65%, 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% increase in small tissue remnants less than 5 $ft^2$ (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 4 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).

Overall, it appears that populations of mountainous star coral have been decreasing. Population decline has occurred over the past few decades with a 65% loss in mountainous star coral cover across 5 countries. Losses of mountainous star coral from Mona and Descheo Islands, Puerto Rico include a 36-48% reduction in abundance and a decrease of 42-59% 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.

### 3.6.4 Threats

Detailed information on the threats to mountainous star coral can be found in the Final Listing Rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Mountainous star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

Mountainous star coral is highly susceptible to elevated temperatures. In lab experiments, elevated temperatures resulted in misshapen embryos and differential gene expression in larvae that could indicate negative effects on larval development and survival. Bleaching susceptibility is generally high; 37-100% of mountainous star coral colonies have reported to bleach during several bleaching events. Chronic local stressors can exacerbate the effects of warming temperatures, which can result in slower recovery from bleaching, reduced calcification, and slower growth rates for several years following bleaching. Additionally, disease outbreaks affecting mountainous star coral have been linked to elevated temperature as they have occurred after bleaching events. We conclude that mountainous star coral is highly susceptible to elevated temperature.

Surveys at an inshore patch reef in the Florida Keys that experienced temperatures less than 18˚C for 11 days revealed species-specific cold-water susceptibility and low survivorship. Mountainous star coral was one of the more susceptible species with 90% of colonies experiencing total colony mortality, including some colonies estimated to be more than 200 years old (Kemp et al. 2011). In surveys from Martin County to the lower Florida Keys, mountainous star coral was the second most susceptible coral species, experiencing an average of 37% partial mortality (Lirman et al. 2011).

Mountainous star coral is highly susceptible to ocean acidification. Laboratory studies indicate that ocean acidification affects that mountainous star coral both through reduced fertilization of gametes and reduced growth of colonies (Carricart-Ganivet et al. 2012).

Mountainous star coral is often among the coral species with the highest disease prevalence and tissue loss. Outbreaks have been reported to affect 10-19% of mountainous star coral colonies, and yellow band disease and white plague have the greatest effect. Disease often affects larger colonies, and reported tissue loss due to disease ranges from 5-90%. Additionally, yellow band disease results in lower fecundity in diseased and recovered colonies of mountainous star coral. Therefore, we anticipate that mountainous star coral is highly susceptible to disease.

Sedimentation can cause partial mortality of mountainous star coral, and genus-level information indicates that sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance. Therefore, we anticipate that mountainous star coral is highly susceptible to sedimentation.

Although there is no species-specific information, the star coral species complex is susceptible to nutrient enrichment through reduced growth rates, lowered recruitment, and increased disease severity. Therefore, based on genus-level information, we anticipate that mountainous star coral is likely highly susceptible to nutrient enrichment.

### 3.6.5 Summary of Status

Mountainous star coral has undergone major declines mostly due to warming-induced bleaching and disease. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and reduced thermal tolerance due to chronic local stressors stemming from land-based sources of pollution. Mountainous star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate its 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. The buffering capacity of these life history characteristics, however, is expected to decrease as colonies shift to smaller size classes as has been observed in locations in 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 mountainous star coral is limited to an area with high, localized human impacts and predicted increasing threats. Its depth range of 0.5 m to at least 40 m, possibly up 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, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than those in which the species occurs. Mountainous star coral occurs in most reef habitats, including both 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 temperatures and ocean chemistry at any given point in time. Its abundance, life history characteristics, and depth distribution, 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. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

## 3.7 Status of Lobed Star Coral

On September 10, 2014, NMFS listed lobed star coral as threatened (79 FR 53852).

### 3.7.1 Species Description and Distribution

Lobed star coral colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other 2 star coral species, margins on the sides of columns are typically dead. Live colony surfaces usually lack ridges or bumps.

Lobed star coral is common throughout the western Atlantic Ocean and greater Caribbean Sea including the Flower Garden Banks, but may be absent from Bermuda. Lobed star coral is reported from most reef environments in depths of approximately 1.5-66 ft (0.5-20 m). The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >100 ft [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.

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% (thus, 14-82% 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.

### 3.7.2 Life History Information

The star coral species complex has growth rates ranging from 0.02-0.5 in (0.06-1.2 cm) per year and averaging approximately 0.3 in (1 cm) 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 more slowly in deeper water and in less clear water.

All 3 species of the star coral complex are hermaphroditic broadcast spawners[13], with spawning concentrated on 6-8 nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All 3 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 1-2 hours earlier. Fertilization success measured in the field was generally below 15% for all 3

[13] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

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 2 species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 12 $in^2$ (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 130 $ft^2$ (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, lobed star corals 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 lobed 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.

### 3.7.3 Status of Population Dynamics

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 two years after the bleaching event, but they returned and then stabilized at the lower rate the following year.

Colony density varies by habitat and location, and ranges from less than 0.1 to greater than 1 colony per approximately 100 $ft^2$ (10 $m^2$). Benthic surveys along the Florida Reef Tract between 1999 and 2017 recorded an average density of 0.01 to 0.09 colonies per $m^2$, and lobed star coral was observed at 4% to 16% of surveyed sites (NOAA, unpublished data). Average density of lobed star corals in Puerto Rico ranged from 0.01 to 0.08 colonies per $m^2$ in surveys conducted between 2008 and 2018 and was observed at 9% to 63% of surveyed sites (NOAA, unpublished data). In the U.S. Virgin Islands, average density ranged from 0.03 to 0.21 colonies per $m^2$ in benthic surveys conducted between 2002 and 2017, and lobed star coral was observed at 25% to 54% of surveyed sites (NOAA, unpublished data). In the Flower Garden Banks, limited surveys detected lobed star corals at none to 24% of surveyed sites, and density was recorded as 0.1 colonies per $m^2$ in 2010 and 0.01 colonies per $m^2$ in 2013 (NOAA, unpublished data). Off southwest Cuba on remote reefs, average lobed star coral density was 0.31 colonies per approximately 108 $ft^2$ (10 $m^2$) at 38 reef-crest sites and 1.58 colonies per approximately 108 $ft^2$ (10 $m^2$) at 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. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016). Lobed star coral was one of the species in surveys that showed the highest prevalence of disease, and populations were reduced to < 25% of the initial population size (Precht et al. 2016).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 43-44% of lobed star corals were impacted (NOAA 2018). In Florida, approximately 80% 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 U.S. Virgin Islands though qualitative observations indicate that damage was also 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 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% in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008a). Cover of lobed star coral declined 71% in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Star corals are the 3rd most abundant coral by percent cover in permanent monitoring stations in the U.S. Virgin Islands. A decline of 60% was observed between 2001 and 2012 primarily due to bleaching in 2005. However, most of the mortality was partial mortality, and colony density in monitoring stations did not change (Smith 2013).

Bruckner and Hill (2009) did not note any extirpation of lobed star coral at 9 sites off Mona and Desecheo Islands, Puerto Rico, monitored between 1995 and 2008. However, mountainous star coral and lobed star coral sustained the largest losses with the number of colonies of lobed star coral decreasing by 19% and 20% at Mona and Desecheo Islands, respectively. In 1998, 8% of all corals at 6 sites surveyed off Mona Island were lobed star coral colonies, dipping to approximately 6% in 2008. At Desecheo Island, 14% of all coral colonies were lobed star coral in 2000 while 13% were in 2008 (Bruckner and Hill 2009).

In a survey of 185 sites in 5 countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) in 2010 and 2011, size of lobed star coral and boulder star coral colonies was significantly smaller than mountainous star coral. Total mean partial mortality of lobed star coral colonies at all sites was 40%. Overall, the total area occupied by live lobed star coral declined by a mean of 51%, and mean colony size declined from 299 $in^2$ to 146 $in^2$ (1927 $cm^2$ to 939 $cm^2$). There was a 211% increase in small tissue remnants less than 78 $in^2$ (500 $cm^2$), while the proportion of completely live large (1.6-32 $ft^2$ [1,500- 30,000 $cm^2$]) colonies declined. Star coral colonies in Puerto Rico were much larger with large amounts of dead sections. In contrast, colonies in Bonaire were also large with greater amounts of live tissue. The presence of dead sections 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 algal lawns (Bruckner 2012).

Cover of lobed star coral at Yawzi Point, St. John, U.S. Virgin Islands declined from 41% in 1988 to approximately 12% by 2003 as a rapid decline began with the aftermath of Hurricane Hugo in 1989 (Edmunds and Elahi 2007). This decline continued between 1994 and 1999 during a time of 2 hurricanes (1995) and a year of unusually high sea temperature (1998), but percent cover remained statistically unchanged between 1999 and 2003. Colony abundances declined from 47 to 20 colonies per approximately 10 $ft^2$ (1 $m^2$) between 1988 and 2003, due mostly to the death and fission of medium-to-large colonies ($\geq$ 24 $in^2$ [151 $cm^2$]). Meanwhile, the population size class structure shifted between 1988 and 2003 to a higher proportion of smaller colonies in 2003 (60% less than 7 $in^2$ [50 $cm^2$] in 1988 versus 70% in 2003) and lower proportion of large colonies (6% greater than 39 $in^2$ [250 $cm^2$] in 1988 versus 3% in 2003). The changes in population size structure indicated a population decline coincident with the period of apparent stable coral cover. Population modeling forecasted the 1988 size structure would not be reestablished by recruitment and a strong likelihood of extirpation of lobed star coral at this site within 50 years (Edmunds and Elahi 2007).

Lobed star coral colonies were monitored between 2001 and 2009 at Culebra Island, Puerto Rico. The population was in demographic equilibrium (high rates of survival and stasis) before the 2005 bleaching event, but it suffered a significant decline in growth rate (mortality and shrinkage) for 2 consecutive years after the bleaching event. Partial tissue mortality due to bleaching caused dramatic colony fragmentation that resulted in a population made up almost entirely of small colonies by 2007 (97% were less than 7 $in^2$ [50 $cm^2$]). Three years after the bleaching event, the population stabilized at about half of the previous level, with fewer medium-to-large size colonies and more smaller colonies (Hernandez-Delgado et al. 2011b).

Lobed star coral was historically considered to be one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined by 37% to 90% over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. Although star coral remains common in occurrence, abundance has decreased in some areas by 19% to 57%, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, U.S. Virgin Islands, 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. Although lobed star coral is still common throughout the Caribbean, substantial population decline has occurred. 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. Population abundance is likely to decrease in the future with increasing threats.

### 3.7.4 Threats

Detailed information on the threats to lobed star coral can be found in the Final Listing Rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Lobed star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

Lobed star coral is highly susceptible to bleaching with 45-100% of colonies observed to bleach. Reported mortality from bleaching ranges from 2-71%. Recovery after bleaching is slow with paled colonies observed for up to a year. Reproductive failure can occur a year after bleaching, and reduced reproduction has been observed 2 years post-bleaching. There is indication that new algal symbiotic species establishment can occur prior to, during, and after bleaching events and results in bleaching resistance in individual colonies. Thus, lobed star coral is highly susceptible to ocean warming.

In a 2010 cold-water event that affected south Florida, mortality of lobed star coral was higher than any other coral species in surveys from Martin County to the lower Florida Keys. Average partial mortality was 56% during the cold-water event compared to 0.3% from 2005 to 2009. Surveys at a Florida Keys inshore patch reef, which experienced temperatures less than 18˚C for 11 days, revealed lobed star coral was one of the most susceptible coral species with all colonies experiencing total colony mortality.

Although there is no species-specific information on the susceptibility of lobed star coral to ocean acidification, genus information indicates the species complex has reduced growth and fertilization success under acidic conditions. Thus, we conclude lobed star coral likely has high susceptibility to ocean acidification.

Lobed star coral is highly susceptible to disease. Most studies report lobed star coral as among the species with the highest disease prevalence. Disease can cause extensive loss in coral cover, high levels of partial colony mortality, and changes in the relative proportions of smaller and larger colonies, particularly when outbreaks occur after bleaching events.

Lobed star coral has high susceptibility to sedimentation. Sedimentation can cause partial mortality and decreased coral cover of lobed star coral. In addition, genus information indicates sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance. Lobed star coral also has high susceptibility to nutrients. Elevated nutrients cause increased disease severity in lobed star coral. Genus-level information indicates elevated nutrients also cause reduced growth rates and lowered recruitment.

### 3.7.5 Summary of Status

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. 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 areas with high 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. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

## 3.8 Elkhorn and Staghorn Critical Habitat

The term "critical habitat" is defined in Section 3(5)(A) of the ESA as (i) the specific areas within the geographic area occupied by a species, at the time it is listed in accordance with the Act, on which are found those physical or biological features (1) essential to the conservation of the species and (2) that may require special management considerations or protection; and (ii) specific areas outside the geographic area occupied by a species at the time it is listed, upon a determination that such areas are essential for the conservation of the species. "Conservation" is defined in Section 3(3) of the ESA as "…the use of all methods and procedures that are necessary to bring any endangered or threatened species to the point at which listing under the ESA is no longer necessary."

On November 26, 2008, a Final Rule designating elkhorn and staghorn 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 elkhorn and staghorn coral (also known as the essential feature) is substrate of suitable quality and availability in water depths from the mean high water line to 30 meters (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 (50 CFR 226.16(a)). Areas containing this feature have been identified in 4 locations within the jurisdiction of the United States: the Florida area, which comprises approximately 1,329 square miles ($mi^2$) (3,442 square kilometers [$km^2$])

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 $km^2$).

The essential feature 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 essential feature. 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, increases in the dominance of algae since the 1980s impedes coral recruitment. The overexploitation of grazers through fishing has also contributed fleshy macroalgae to persist 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). Monitoring data from the USVI Territorial Coral Reef Monitoring Program indicate that the 2005 coral bleaching event caused the largest documented loss of coral in USVI since coral monitoring data have been available with a decline of at least 50% of coral cover in waters less than 25 m deep ((Smith et al.) 2011). Many of the shallow water coral monitoring stations showed at most a 12% recovery of coral cover by 2011, 6 years after the loss of coral cover due to the bleaching event ((Smith et al.) 2011). The lack of coral cover has led to increases in algal cover on area hard bottom, including the critical habitat essential feature.

# 4 ENVIRONMENTAL BASELINE

This section describes the effects of past and ongoing human and natural factors contributing to the current status of the species, their habitat, and ecosystem within the action area. The environmental baseline describes the species and critical habitat's health based on information available at the time of this consultation.

By regulation, environmental baselines for Opinions include the past and present impacts of all state, federal, or private actions and other human activities in, or having effects in, the action area. We identify the anticipated impacts of all proposed federal projects in the specific action area of the consultation at issue that have already undergone formal or early Section 7 consultation (as defined in 50 CFR 402.11), as well as the impact of state or private actions, or the impacts of natural phenomena, which are concurrent with the consultation in process (50 CFR 402.02).

Focusing on the current state of ESA-listed corals and critical habitat is important because in some areas ESA-listed corals and critical habitat features will commonly exhibit, or be more susceptible to, adverse responses to stressors than they would be in other areas, or may have been exposed to unique or disproportionate stresses. These localized stress responses or stressed baseline conditions may increase the severity of the adverse effects expected from the proposed action.

### 4.1 Status of ESA Listed Corals within the Action Area

The proposed dock is situated north of Blasbalg Point on the south side of Great Cruz Bay. A 2019 benthic study showed a rocky shoreline consisting of coral colonized bedrock extending between 80 ft and 100 ft offshore, giving way to a narrow sand belt before dense submerged aquatic vegetation (SAV) consisting of 50-70% patchy seagrass begin farther out into the bay. There are coral colonized boulders and coral heads scattered within the sand belt where some of the corals which colonize the scattered boulders are ESA-listed species. Elkhorn (*A. palmata*) and mountainous star coral (*O. faveolata*) were found to occur at the border of the impact area of the project footprint, with the elkhorn coral having colonized the footprint of the previous dock (Figure 4).

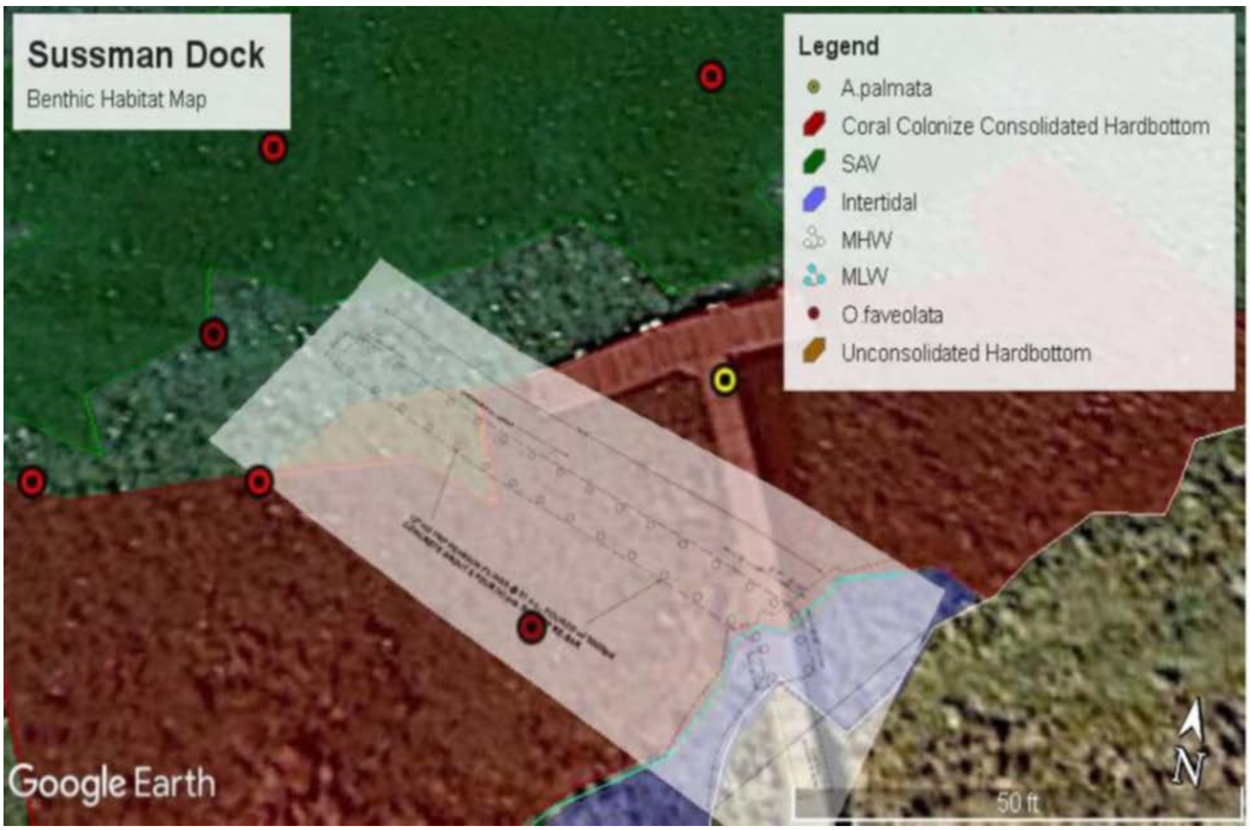


**Figure 4. Benthic habitat map relative to the proposed dock. Habitat map superimposed on pre-hurricane image. Image provided by the Action Agency.**

## 4.2 Status of Designated Critical Habitat within the Action Area

### 4.2.1 Elkhorn and Staghorn Critical Habitat

This Opinion focuses on an activity occurring within the St. Thomas/St. John elkhorn and staghorn critical habitat unit, which comprises approximately 121 square miles ($mi^2$) or 77,440 acres (ac) 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 National Ocean Service's (NOS) Biogeography Program in 2000 (Kendall et al. 2001). The most recent benthic survey conducted in 2019 reported unconsolidated and consolidated hardbottom within the action area, including all ESA-listed present in the habitat surrounding the proposed dock, non-ESA-listed corals and seagrass.

## 4.3 Factors Affecting all ESA-Listed Corals and Critical Habitat within the Action Area

### 4.3.1 Federal Actions

A search of NMFS's records found no projects in the action area that have undergone Section 7 consultation, or have had effects within the action area, as per a review of the NMFS Protected Resource Division (PRD) completed consultation database by the consulting biologist on June 12, 2020.

### 4.3.2 Other Potential Sources of Impacts to the Environmental Baseline

Although many regulations exist to protect corals, including elkhorn and staghorn corals and their habitat, many of the activities identified as threats still adversely affect the species. Poor boating and anchoring practices, poor snorkeling and diving techniques and adverse fishing practices can cause physical damage to ESA-listed corals and elkhorn and staghorn critical habitat. Commercial and recreational vessel traffic can adversely affect ESA-listed coral colonies and coral critical habitat through propeller scarring, propeller wash, and accidental groundings. Fishing gear such as hook and line and nets can damage corals and coral habitat by entanglement. Fish traps can affect corals and coral habitat if they are moved and dragged onto reefs and marine bottom by storms and waves. Nutrient, contaminants and sediments from non-point source and point source pollution can degrade elkhorn and staghorn critical habitat and nearshore habitats used by sea turtles inducing algal blooms or sediment build up that prohibit coral settlement and growth.

Stochastic events, such as tropical storms and hurricanes, are common throughout the range of elkhorn and staghorn critical habitat. These events are by nature unpredictable and can adversely affect critical habitat through sediment deposition and coral damage. In 2017, Hurricanes Irma and Maria likely damaged habitat in and around the action area.

### 4.3.3 Conservation on Recovery Actions Shaping the Environmental Baseline

The CFMC has established regulations prohibiting the use of bottom-tending fishing gear in certain areas in the federal waters of the Exclusive Economic Zone (EEZ). These areas are either closed to any fishing seasonally or permanently closed to all fishing. The Territory has similar fisheries regulations for both commercial and recreational fishers.

Numerous management mechanisms exist to protect corals or coral reefs in general. Existing federal regulatory mechanisms and conservation initiatives most beneficial to branching corals have focused on addressing physical impacts, including damage from fishing gear, anchoring, and vessel groundings. The Coral Reef Conservation Act and the Magnuson-Stevens Act Coral and Reef Fish Fishery Management Plans (Caribbean) require the protection of corals and prohibit the collection of hard corals. Depending on the specifics of zoning plans and regulations, marine protected areas (MPAs) can help prevent damage from collection, fishing gear, groundings, and anchoring.

The Territorial Government regulates activities that occur in terrestrial and marine habitats of USVI. The V.I. Code prohibits the taking, possession, injury, harassment, sale, offering for sale, etc. of any indigenous species, including live rock (V.I. Code Title 12 and the Indigenous and Endangered Species Act of 1990). Additionally, USVI has a comprehensive, state regulatory program that regulates most land, including upland and wetland, and surface water alterations throughout the Territory, including in partnership with NOAA under the Coastal Zone Management Act, and EPA under the Clean Water Act.

The Coral and Reef Associated Plants and Invertebrates FMP of the CFMC prohibits the extraction, possession, and transportation of any coral, alive or dead, from federal waters unless a permit is issued. Similarly, the CFMC prohibits the use of chemicals, plants, or plant-derived toxins and explosives to harvest coral (50 CFR 622.9). The CFMC also prohibits the use of pots/traps, gill/trammel nets, and bottom longlines on coral or hard bottom year-round in existing seasonally closed areas in the EEZ (50 CFR 622.435).

The final Section 4(d) rule for elkhorn and staghorn corals (73 FR 64264; October 29, 2008) generally applies the prohibitions of section 9(a) of the ESA to elkhorn and staghorn corals, subject to certain exceptions. 50 CFR 223.208(a). Namely, the take of elkhorn and staghorn corals during certain restoration activities, defined as "the methods and processes used to provide aid to injured individuals," when they are conducted by certain federal, state, territorial, or local government agency personnel or their designees acting under existing legal authority, is excepted from the prohibitions in sections 9(a)(1)(B) and (C) of the ESA and may be conducted promptly without the need for an ESA Section 10 incidental take permit. 50 CFR 223.208(c)(2). Restoration activities are also carried out to restore damaged critical habitat.

A recovery team comprised of fishers, scientists, managers, and agency personnel from Florida, Puerto Rico, and USVI, and federal representatives was convened by NMFS and has created a recovery plan based upon the latest and best available information for elkhorn and staghorn corals and their habitat (NMFS 2015).

USVI has a comprehensive, state regulatory program that regulates most land, including upland and wetland, and surface water alterations throughout the Territory, including in partnership with NOAA under the Coastal Zone Management Act, and EPA under the Clean Water Act. The Caribbean Fishery Management Council (CFMC) prohibits the use of chemicals, plants, or plant-derived toxins and explosives to harvest coral (50 CFR 622.9). The CFMC also prohibits the use of pots/traps, gill/trammel nets, and bottom longlines on coral or hard bottom year-round in existing seasonally closed areas in the EEZ (50 CFR 622.435).

Critical habitat for ESA-listed elkhorn and staghorn corals was designated through a final rule published in 2008. The critical habitat designation requires federal agencies consult on actions may adversely affect critical habitat to ensure that the actions do not result in adverse modification or destruction of the critical habitat. This reduces the threats to elkhorn and staghorn corals by adding a layer of protection to habitat necessary for the conservation of the species.

The NOAA Coral Reef Conservation Program, through its internal grants, external grants, and grants to the Territory and the CFMC, has provided funding for several activities with an education and outreach component for informing the public about the importance of the coral reef ecosystem of the USVI. The Southeast Regional Office of NMFS has also developed outreach materials regarding the conservation of all ESA-listed corals, and the designation of coral critical habitat. These materials have been circulated to constituents during education and outreach activities and public meetings, and as part of other Section 7 consultations, and are readily available on the web at: https://www.fisheries.noaa.gov/.

# 5 EFFECTS OF THE ACTION ON ESA-LISTED CORALS AND CRITICAL HABITAT

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 (50 CFR 402.02).

In this section of our Opinion, we assess the effects of the action on ESA-listed species and designated critical habitat that are likely to be adversely affected. The analysis in this section forms the foundation for our destruction and adverse medication and jeopardy analyses in Sections 7 and 8.

## 5.1 Effects of the Action on Elkhorn and Staghorn Coral Designated Critical Habitat

As described herein, NMFS believes that the proposed action is likely to adversely affect elkhorn and staghorn coral designated critical habitat within the St. Thomas/St. John area. As part of this Opinion and because the action will result in adverse effects to elkhorn and staghorn coral critical habitat, NMFS must evaluate whether the action is likely to result in destruction or adverse modification of critical habitat. If so, NMFS must develop RPAs to avoid the destruction or adverse modification.

The substrate of suitable quality and availability essential feature of elkhorn and staghorn coral critical habitat may be affected by the installation of the following 12-in diameter concrete piles:

- 7 piles installed in sandy substrate,
- 6 piles installed in unconsolidated hardbottom,
- 19 piles installed in consolidated hardbottom, and
- 5 piles installed in un-colonized, intertidal hardbottom.

Of these 37 concrete piles installed, we believe that only 19 piles installed in consolidated hardbottom will affect the substrate of suitable quality and availability essential feature of the elkhorn and staghorn coral designated critical habitat. There is a total of 121 $mi^2$ (77,440 ac)[14] of substrate of suitable quality and availability for elkhorn and staghorn coral designated critical habitat in St. Thomas/St. John area, of which 15 $ft^2$ (0.00034 ac) will be impacted by the installation of 19 piles during dock construction.[15] Thus, we believe the proposed action will adversely affect 15 $ft^2$ of elkhorn and staghorn coral designated critical habitat.

### 5.2 Effects of Proposed Action on ESA-Listed Coral Species

The only identified routes of effect for ESA-listed corals are from the proposed collection and propagation of pillar coral, and out-planting of elkhorn, staghorn, lobed star, mountainous star, and pillar coral colonies.

There are no effects associated with the proposed action that are expected to occur later in time and likely to adversely affect listed species. Effects occurring later in time include redepositing of sediment that may be entrained into the water column from the installation of concrete piles after such sediments have dispersed by currents and waves. The proposed action analyzed in this Opinion is not expected to affect the water column or benthic habitat in any appreciable way other than the impacts described below. The construction of the residential dock is not expected to result in any ongoing effects to the water column or benthic habitat once completed.

The applicant proposes to out-plant up to 200 ESA-listed corals consisting of a combination of elkhorn, staghorn, mountainous star, lobed star, and pillar corals. Pillar coral will be the only coral of opportunity that will be collected for this project to be reared at the TNC nursery for out-planting. The other corals will be provided by TNC from existing stocks. The applicant intends to collect 2 pillar corals of opportunity fragments in order to out-plant up to 40 pillar coral colonies to the out-plant site. When ready for out-planting, up to 200 ESA-listed corals (consisting of elkhorn coral, staghorn coral, mountainous star coral, lobed star coral and pillar coral) will be taken from TNC's coral nursery to the mitigation site and attached with epoxy or cement. This out-planting is to mitigate for the loss of 15 $ft^2$ of elkhorn and staghorn coral critical habitat as a result of pile driving associated with the proposed action.

We believe the proposed mitigation measures would help compensate for the loss of elkhorn and staghorn critical habitat due to project construction. We believe that this portion of the mitigation proposal would have a beneficial effect on designated critical habitat by accelerating

[14] 1 $mi^2$ = 640 ac; therefore 121 $mi^2$ x 640 = 77,440 ac
[15] 15 $ft^2$ x (0.000022957 ac / 1 $ft^2$) = 0.00034 ac

the provision of its intended conservations function. The following analysis shows how we determined that the propagation and out-planting component of the project would provide for the conservation of the species.

Facilitating increased incidence of successful sexual and asexual reproduction is the key objective to the conservation of elkhorn and staghorn corals, as well as the other ESA-listed corals, based on the species' life history characteristics, population declines, and extremely low recruitment (73 FR 72224, November 26, 2008). Therefore, the critical habitat designation for elkhorn and staghorn corals identifies the essential feature within the areas occupied by the species that need protection to support that goal. Corals are sessile and depend upon external fertilization in order to produce larvae. Fertilization success is reduced as adult density declines (known as the Allee effect) (Levitan 1991). Since elkhorn, staghorn and pillar corals are not able to self-fertilize they require a certain density (discussed in further detail below) of adult colonies to promote sexual reproduction (*Acropora* Biological Review Team 2005, Section 4.3.2 Life History Information).

Another activity that supports the goal of increased incidence of successful sexual and asexual reproduction is artificial propagation of the species. The Recovery Plan for Elkhorn and Staghorn Coral (NMFS 2015) identifies the following key action necessary to promote conservation:

*Develop and implement appropriate strategies for population enhancement, through restocking and active management, in the short to medium term, to increase the likelihood of successful sexual reproduction and to increase wild populations.*

The collection and propagation of pillar coral, and out-planting of elkhorn, staghorn, lobed star, mountainous star, and pillar corals at a natural and existing coral mitigation site may result in some adverse effects to those corals. While corals may experience some stress from being physically moved from one location to another, coral specialists have developed transfer techniques that have resulted in managing the stress on corals with non-lethal effects (Bowden-Kerby, 2014). Therefore, NMFS does not expect that coral collection and propagation will result in lethal take. Additionally, there will be beneficial effects because it will enhance species recovery by establishing wild populations that are poised to reproduce sexually and asexually, which is achieving the conservation objective of designated critical habitat.

Corals of opportunity occur from storm events and groundings that dislodge parts of a colony and they fall to the substrate. They may remain there unattached and continue to survive for a period. However, reattached coral fragments show significantly high rates of survival, as compared to fragments that are left unattached, due to burial by sediment, part of the fragment being suffocated from laying on the side, and from abrasion from being moved around by waves and currents (Griffin et al. 2015; Lirman 2000). This stress from being unattached reduces the fragment's chances of survival. Although collecting and reattaching corals of opportunity may result in some adverse effects, this action will be beneficial overall because it will substantially increase the chances of fragment survival.

We believe that coral collection will result in non-lethal take of 2 pillar coral fragments, and that coral outplanting will result in non-lethal take of up to 40 pillar coral colonies and up to 160 coral colonies consisting of one or more elkhorn, staghorn, mountainous star, and/or lobed star coral colonies (the actual number of each coral colony to be outplanted will be determined at the time of outplant and based on the availability of these species in the nursery at the time). In summary, we believe that the proposed mitigation measure would in fact compensate for the loss of 15 $ft^2$ of elkhorn and staghorn critical habitat and would have a beneficial effect by accelerating the provisions intended for its intended conservation function.

## 6 CUMMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, or local private actions that are reasonably certain to occur in the action areas considered in this Opinion. Future federal actions that are unrelated to the proposed actions are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

Most activities affecting elkhorn and staghorn critical habitat are regulated federally; therefore, any future activities within the action area, which is in waters of the U.S., will likely require ESA Section 7 consultation. However, upland development, whether for housing or agriculture, often has no federal nexus if the project is located on uplands and is small in size. Depending on the number and location of these developments, sediment and nutrient loading to nearshore waters could become a chronic stressor, which would affect elkhorn and staghorn coral critical habitat and ESA-listed corals.

NMFS is not aware of any proposed or anticipated changes in human-related actions (e.g., recreational use, fisheries, habitat degradation including from vessel use) or natural conditions that would substantially change the impacts that each threat has on elkhorn and staghorn coral critical habitat, or any additional future state, tribal, or local private actions that are reasonably certain to occur in the action area in the future beyond the potential development described above. Therefore, NMFS expects that the levels of interactions with elkhorn and staghorn critical habitat and ESA-listed corals described for each of the fisheries and non-fisheries activities in Section 4.3 (Factors Affecting All ESA-Listed Corals and Critical Habitat within the Action Area) will continue at similar levels into the foreseeable future.

## 7 DESTRUCTION/ADVERSE MODIFICATION ANALYSIS

NMFS's regulations define *destruction or adverse modification* to mean "a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species" (50 CFR 402.02). Other alterations that may destroy or adversely modify critical habitat may include impacts to the area itself, such as those that would impede access to or use of the essential features. NMFS will place impacts to critical habitat into the context of the overall designation to determine if the overall value of the critical habitat is likely to be appreciably reduced. While the destruction or adverse modification analysis will consider the nature and significance of effects that occur at a smaller scale than the whole designation, the ultimate determination applies to the value of the critical habitat designation as a whole. The extent to which the proposed action is anticipated to impact the development of some important physical or biological features is a relevant consideration for the Services' critical habitat analysis. Generally, we conclude that a Federal action is likely to "destroy or adversely modify" designated critical habitat if the action results in an alteration of the quantity or quality of the essential physical or biological features of designated critical habitat, or that precludes or significantly delays the capacity of that habitat to develop those features over time, and if the effect of the alteration is to appreciably diminish the value of critical habitat for the conservation of the species.

Ultimately, we seek to determine if, with the implementation of the proposed action, critical habitat would remain functional (or retain the current ability for the essential features to be functionally established) to serve the intended conservation role for the species. This analysis takes into account the geographic and temporal scope of the proposed action, recognizing that "functionality" of critical habitat necessarily means that it must now and must continue in the future to support the conservation of the species and progress toward recovery. Thus, the analysis must take into account any changes in amount, distribution, or characteristics of the critical habitat that will be required over time to support a successfully recovering species. Destruction or adverse modification does not depend strictly on the size or proportion of the area adversely affected, but rather on the role the action area and the affected critical habitat serves with regard to the function of the overall critical habitat designation, and how that role is affected by the action.

### 7.1 Elkhorn and Staghorn Coral Critical Habitat

Critical habitat was designated for elkhorn and staghorn corals, in part, because further declines in the low population sizes of the species could lead to threshold levels that make the chances for recovery low. More specifically, low population sizes for these species could lead to an Allee effect[16] and lower effective density (of genetically distinct adults required for sexual reproduction), and a reduced source of fragments for asexual reproduction and recruitment. Therefore, the key conservation objective of designated critical habitat is to facilitate increased incidence of successful sexual and asexual reproduction, which in turn facilitates increases in the species' abundances, distributions, and genetic diversity. To this end, our analysis of whether the proposed action is likely to destroy or adversely modify designated critical habitat seeks to determine if the adverse effects of the proposed action on the essential feature of designated elkhorn and staghorn critical habitat will appreciably reduce the capability of the critical habitat to facilitate an increased incidence of successful sexual and asexual reproduction. This analysis takes into account the status of the species during the installation of the new dock. This analysis also takes into account the geographic and temporal scope of the proposed action.

An area of 15 $ft^2$ containing the elkhorn and staghorn critical habitat essential feature will be permanently removed due to pile installation. As noted in the critical habitat rule (73 FR 72210, November 26, 2008), the loss of suitable habitat is one of the greatest threats to the recovery of listed coral populations. The loss of suitable habitat affects the reproductive success of listed corals because substrate for sexual recruits to settle is lost. Thus, the value of critical habitat for the conservation of the species is to facilitate an increased incidence of successful sexual and asexual reproduction. Nevertheless, NMFS does not believe the installation 19 12-in diameter

[16] The Allee effect is the effect of population density on population growth by which reproductive rates fall at very low population densities and reproduction and survival of individuals increase as population density increases.

concrete piles will permanently alter the overall suitability or habitat quality of elkhorn and staghorn coral critical habitat in the action area or throughout the critical habitat units, or prevent the critical habitat from facilitating successful sexual and asexual reproduction. Approximately 121 $mi^2$ are likely to contain the essential feature of ESA-designated elkhorn and staghorn coral critical habitat within the St. Thomas/St. John unit, based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOAA's National Ocean Service (NOS) Biogeography Program in 2000 (Kendall et al. 2001a).

There is a total of 121 $mi^2$ (77,440 ac)[17] of designated critical habitat in St. Thomas/St. John area. Impacting approximately 15 $ft^2$ of elkhorn and staghorn coral critical habitat represents approximately 0.00000044% of the essential feature within the St. Thomas/St. John critical habitat unit.[18] Given the very small size (15 $ft^2$) of the impact to hardbottom compared to the area containing elkhorn and staghorn coral critical habitat within the St. Thomas/St. John area for the dock installation project, NMFS does not anticipate that the action area containing the essential feature will cease to function as adequate substrate for settlement of listed coral larvae, reattachment of listed coral fragments, and growth of listed coral colonies. Therefore, NMFS does not believe the installation of a new dock will have an appreciable impact on the ability of elkhorn and staghorn coral critical habitat in the St. Thomas/St. John unit to provide for the conservation of these Acroporid corals.

Based on the above analysis, we conclude that the adverse effects on elkhorn and staghorn coral critical habitat due to the proposed action will not impede the capability of the critical habitat to facilitate an increased incidence of successful sexual and asexual reproduction and, therefore will not appreciably diminish the value of critical habitat for the conservation of the species.

## 8 JEOPARDY ANALYSIS

The analyses conducted in the previous sections of this Opinion serve to provide a basis to determine whether the proposed action is likely to jeopardize the continued existence of ESA-listed corals. In Section 5, we outlined how the proposed action can effect these species. Now we turn to an assessment of the species' response to these impacts, in terms of overall population effects, and whether those effects of the proposed action, when considered in the context of the status of the species (Section 3), the environmental baseline (Section 4), and the cumulative effects (Section 6), will jeopardize the continued existence of the affected species.

This section evaluates whether the proposed action is likely to jeopardize the continued existence of pillar, elkhorn, staghorn, mountainous star, and/or lobed star corals in the wild. To *jeopardize the continued existence of* is defined as "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Thus, in making this determination, NMFS must first determine whether the proposed action directly or indirectly reduce the reproduction, numbers, or

[17] 1 $mi^2$ = 640 ac; therefore 121 $mi^2$ x 640 = 77,440 ac
[18] (0.00034 ac / 77,440 ac) x 100 = 0.00000044 %

distribution of a listed species. Then if there is a reduction in one or more of these elements, we evaluate whether it would be expected to cause an appreciable reduction in the likelihood of both the survival and the recovery of the species in the wild.

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

In the following analysis, we evaluate the effects of the collection of pillar coral and out-planting of ESA-listed corals (including pillar, elkhorn, staghorn, mountainous star, and/or lobed star corals) in the action area.

As discussed in section 5.3, the applicant will collect up to 2 pillar corals of opportunity for the purpose of out-planting to compensate for the loss of elkhorn and staghorn coral critical habitat essential feature caused by pile driving for the new dock. The 2 coral fragments will be propagated at the TNC nursery and up to 40 pillar corals will be out-planted at the designated out-plant site, along with up to 160 ESA-listed corals consisting of elkhorn, staghorn, lobed star, and mountainous star corals. The proposed action is not expected to negatively affect the current geographic range or spatial distribution of pillar corals because we expect that the species will persist within the action area due to out-planting of colonies. It will prevent the mortality of these corals were they left unattached and not collected. This action will be generally beneficial for all of the ESA-listed coral species collected, and will increase the biological diversity within the action area.

We expect that all of the ESA-listed corals subject to out-planting will have a very high survival rate. Numerous nurseries for corals have been established to support this recovery activity in the past 15 years with the expressed purpose of enhancing wild populations with sufficient densities of the species to promote natural sexual reproduction (Johnson et al. 2011). It is our opinion that there may be some stress to corals due to collection and out-planting, however, we believe this will not result in lethal take.

The out-planting may have an increase in the long-term reproduction of the species in the action area. This action will enhance and benefit all ESA-listed coral species to be out-planted by preventing mortality and by increasing their abundance, reproduction and distribution. Therefore, NMFS believes that the proposed restoration is not likely to reduce the likelihood of staghorn coral, elkhorn coral, mountainous star coral, lobed star coral, and pillar corals' survival or recovery in the wild.

## 9 CONCLUSION

After reviewing the current status of elkhorn and staghorn designated critical habitat, the environmental baseline, and the cumulative effects, it is our opinion that the loss of 15 $ft^2$ (0.00034 ac) from the proposed action will not impede the critical habitat's ability to support the conservation of the species, despite permanent adverse effects. Therefore, we conclude that the action, as proposed, is likely to adversely affect, but is not likely to destroy or adversely modify, elkhorn and staghorn designated critical habitat. It is also our Opinion that the proposed action is *not* likely to jeopardize the continued existence of pillar coral from coral fragment collection and out-planting. Nor is the proposed action likely to jeopardize the continued existence of elkhorn, staghorn, lobed star, and mountainous star corals from out-planting.

## 10 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and federal regulation pursuant to Section 4(d) of the ESA prohibit take of endangered and threatened species, respectively, without special exemption. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement (ITS).

*Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. NMFS must estimate the type and extent of incidental take expected to occur from implementation of the proposed action to frame the limits of the take exemption provided in the Incidental Take Statement. These limits set thresholds that, if exceeded, would be the basis for reinitiating consultation. The following section describes the type and extent of take that NMFS anticipates will occur as a result of implementing the proposed action, and on which NMFS has based its determination that the action is not likely to jeopardize listed species.

As noted above, the take of elkhorn coral and staghorn coral is generally prohibited by 50 CFR 223.208(a), promulgated pursuant to section 4(d) of the ESA. However, the rule provides an exception from the prohibitions in sections 9(a)(1)(B) and (C) of the ESA for the take of elkhorn coral and staghorn coral that occurs during certain restoration activities. 50 CFR 223.208(c)(2). The proposed outplanting of elkhorn and staghorn coral colonies will use methods and processes used to provide aid to injured individuals, and thus qualifies as a restoration activity that is excepted from take prohibitions in sections 9(a)(1)(B) and (C) of the ESA. Additionally, the take of pillar, mountainous star, and lobed star corals is not prohibited, as NMFS has not promulgated a Section 4(d) rule for these species (79 FR 53852, Publication Date September 10, 2014).

As a result, the take of ESA-listed corals as part of the proposed action is not prohibited by the ESA. However, a circuit court case held that non-prohibited incidental take must be included in the ITS.[19] Providing an exemption from Section 9 liability is not the only purpose of specifying take in an incidental take statement. Specifying incidental take ensures we have a metric against which we can measure whether or not reinitiation of consultation is required. It also ensures that we identify reasonable and prudent measures that we believe are necessary or appropriate to minimize the impact of such incidental take.

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

### 10.1 Amount or Extent of Take

NMFS has determined that the proposed project will result in the non-lethal take of up to:

- 2 fragments of pillar coral during collection
- 40 total pillar coral fragments during out-planting
- 160 total ESA-listed coral colonies consisting of one or more elkhorn, staghorn, mountainous star, and/or lobed star coral colonies during out-planting.

### 10.2 Effects of the Take

NMFS has determined the anticipated level of incidental take is not likely to jeopardize the continued existence of the species identified above.

## 11 REASONABLE AND PRUDENT MEASURES

Section 7(b)(4) of the ESA requires NMFS to issue to any agency whose proposed action is found to comply with Section 7(a)(2) of the ESA, but may incidentally take individuals of listed species, a statement specifying the impact of that taking. It also requires NMFS to identify RPMs necessary to minimize the impacts from the agency action, and terms and conditions to implement those measures. Only incidental taking by the federal agency or applicant that complies with the specified terms and conditions is authorized.

The RPMs and terms and conditions are required, per 50 CFR 402.14 (i)(1)(ii) and (iv), to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. These measures, terms and conditions are nondiscretionary, and must be implemented by the USACE or the applicant in order for the protection of Section 7(o)(2) to apply. The USACE has a continuing duty to regulate the activity covered by this ITS. If the USACE or the applicant fails to adhere to the terms and conditions of the ITS through

[19] *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012). Though the *Salazar* case is not a binding precedent for this action, which occurs outside of the Ninth Circuit, we find the reasoning persuasive and are following the case out of an abundance of caution and in anticipation that the ruling will be more broadly followed in future cases.

enforceable terms, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse. To monitor the impact of the incidental take, the USACE or the contractor must report the progress of the action and its impact on the species to NMFS as specified in the ITS [50 CFR 402.12(i)(3)].

NMFS has determined that the following RPM is necessary or appropriate to minimize impacts of the incidental take of elkhorn coral, staghorn coral, mountainous star coral, lobed star coral, and pillar corals during the proposed action.

1. USACE must record and maintain the status and disposition of all ESA-listed species specified in the Incidental Take Statement.
2. The USACE must conduct and document biological and environmental monitoring.

### 11.1 Terms and Conditions

The USACE must comply with the following terms and conditions, which implement the RPM described above (50 CFR 402.14). These terms and conditions are nondiscretionary.

1. USACE must inventory and track the location, health, and size of all collected pillar coral colonies. (RPM1)

2. USACE must record the location of all out-planted coral colonies specified in the ITS. (RPM 1)

3. USACE shall submit copies of all mitigation and monitoring reports to NMFS at the letterhead address. The USACE must provide NMFS with all data collected during monitoring events conducted, as well as any monitoring reports generated following the completion of the proposed project. The monitoring programs shall include reporting requirements to ensure NMFS, USACE, and other relevant agencies are aware of corrective actions being taken when thresholds are exceeded, as well as ensure NMFS receives data related to the condition of listed corals in the area due to the importance of these listed species. (RPM 2)

The RPM, with its implementing terms and conditions, is designed to minimize the impact of incidental take that might otherwise result from the implementation of the RPA. If, during the course of the action, this level of incidental take is exceeded, such incidental take represents new information requiring reinitiation of consultation and review of the RPM and its implementing terms and conditions. The USACE must immediately provide an explanation of the cause(s) of the take exceedance and review with NMFS the need for possible modification of the RPM and its implementing terms and conditions.

## 12 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species.  Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

We believe the following conservation recommendations further the conservation of ESA-listed Nassau grouper, corals, and staghorn and elkhorn coral designated critical habitat.  We strongly recommend consideration and adoption of these measures.  In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, we request notification of the implementation of any conservation recommendations.

1. Provide NMFS Southeast PRD with copies of all monitoring reports for coral out-planting.

2. We recommend that pre, during, and post-construction surveys include surveys for Nassau grouper, and that any sighting of this species be reported to NMFS so that we can update information related to the presence of the species throughout its range.

Please notify NMFS if the federal action agency carries out any of these recommendations so that we will be kept informed of actions that are intended to improve the conservation of listed species or their designated critical habitats.

## 13 REINITIATION OF CONSULTATION

This concludes formal consultation on the proposed action.  As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of taking specified in the proposed action is exceeded; (2) new information reveals effects of the action that may affect 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 listed species or critical habitat that was not considered in the Biological Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action.  In instances where the amount or extent of incidental take is exceeded, the USACE must immediately request reinitiation of formal consultation and project activities may only resume if the USACE establishes that such continuation will not violate sections 7(a)(2) and 7(d) of the ESA.

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